NOAA Technical Memorandum NOS OR&R 1

Monitoring of Biological Recovery of Prince William Sound Intertidal Sites Impacted by the *Exxon Valdez* Oil Spill

1997 Biological Monitoring Survey

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1997 Biological Monitoring Survey

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EXECUTIVE SUMMARY

Marine organisms residing along the shoreline of Prince William Sound were impacted by exposure to oil spilled from the T/V *Exxon Valdez* in 1989. They also experienced mechanical disturbance during invasive oil-spill remediation and cleanup efforts that were conducted along portions of the impacted shoreline. The impacts to intertidal biota and their subsequent recovery are evident in population data collected during annual surveys conducted in the Sound between 1989 and 1997. Although there are many aspects of recovery, this report focuses on population increases that occurred at impacted sites over a period of one to two years beginning around 1990. Recolonization occurred across the full range of intertidal assemblages, including sediment-dwelling infaunal invertebrates, sessile and motile epifaunal invertebrates, and algae. It was also evident at all intertidal elevations sampled. For the most part, population increases at impacted sites were statistically significant ($p \leq 0.10$) compared to population fluctuations observed at nonoiled control sites. After the initial increase, intertidal populations at impacted sites stabilized with abundance perturbations that tracked those of control sites, albeit with different average abundance levels. Thus, within the resolution of statistical tests applied to abundance, the major intertidal assemblages had largely recolonized impacted sites and had achieved equilibrium with ambient environmental conditions by 1993.

The amplitude of the population increase was larger at oiled sites that were subjected to aggressive cleanup techniques. In part, this result is consistent with past observations of increased damage to intertidal organisms at sites treated with high-pressure hot-water washes. The enhanced recolonization at these sites reflects the increased damage. Also, there was no evidence that recovery was measurably delayed at the oiled sites that received hot-water washing. In fact, timing and duration of the repopulation was remarkably similar across the various impacted sites, tidal elevations, and intertidal assemblages.

The timing and degree of recovery was determined from a comparison of intertidal abundance at three categories of sites:

- Category 1 - nonoiled control sites;
- Category 2 - sites that were oiled but not hot-water washed, but may have received light cleaning; and
- Category 3 - sites along shorelines that were washed with high-pressure hot water to remove surficial oil deposits.

In this report, recovery assessments were based on logarithmically transformed abundance data to characterize the multiplicative effects of mortality (impacts) and natality (recovery). On the logarithmic scale, impacts from oiling and shoreline cleanup are conspicuous as an initially low abundance at impacted sites relative to control populations (Figure 1). While this Figure shows time profiles of sediment-dwelling biota (infauna), epifaunal invertebrates and algae exhibited a similar signature of recovery. Control sites maintained a relatively stable abundance level throughout monitoring whereas the low incipient abundance at impacted sites in 1990 and 1991 recovered rapidly (within a year) to a population level that remained comparatively stable throughout the six subsequent years.

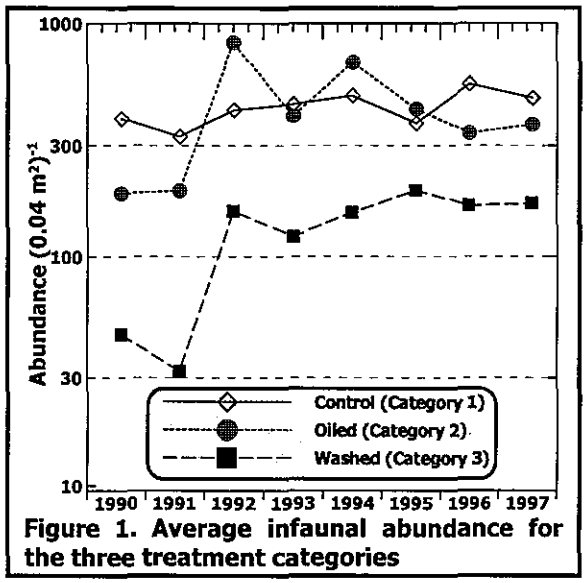


Figure 1. Average infaunal abundance for the three treatment categories

Infaunal Recovery

Figure 1 shows that the initial infaunal abundance at untreated oiled sites (Category 2) was not as low as at hot-water washed sites (Category 3). Nevertheless, an abrupt (intra-annual) increase in total abundance occurred simultaneously at both categories of impacted sites. After two years of low infaunal abundance in 1990 and 1991, the steep rise in population took less than a year and was complete by the 1992 survey. The significance of this recovery process is reflected in tests for parallel trends in abundance between control and impacted sites. Parallelism was rejected ($p<0.10$) only within an initial time window (1990-1995) that encompassed the low abundance levels in 1990 and 1991 at washed (Category-3) sites. Significant departures from parallelism were not found between untreated oiled sites (Category 2) and control sites. The inability of the parallelism test to detect ($p\geq0.19$) the smaller-magnitude recovery at Category-2 sites is probably an artifact of the low statistical power associated with the hypothesis test, which had only 4 degrees-of-freedom per six-year test window. Most of the major taxonomic groups (mollusks and marine worms) exhibited a similar distinct pattern of recovery where impacted sites exhibited a sharp increase in abundance relative to control sites followed by a comparatively stable population.

A number of individual taxa participated in the recovery at washed (Category-3) sites although some of these taxa did not stabilize until 1994. These disparate taxa were not necessarily the

most abundant, but exhibited a population trend that significantly departed from that of control sites. The taxa whose abundance increased the most included a small (≤ 1 cm) shrimp-like crustacean (*Cumella vulgaris*); a small shallow-water detritus-feeding gastropod (*Fartulum occidentale*); and three polychaete worms. The major recolonizing polychaetes were also taxonomically distinct and included the leafy paddle worm (*Eteone longa*), which are active predators; the deposit-feeding burrower *Ophelia limacina*; and the highly motile carnivore *Syllis alternata.*

At the Category-2 sites, which were not subjected to high-pressure hydraulic washing, two other polychaete worms recovered the most. These tube-builders included the filter-feeding *Fabriciola berkeleyi* and the terebellid *Laphania boecki.* Another polychaete that contributed to recovery at both categories of impacted sites was the deposit-feeding burrower *Ophelia limacina.* The same was true of the gastropod *Fartulum occidentale*, which experienced a significant increase in its abundance at Category-2 sites compared to controls.

After the initial recovery, total infaunal abundance at washed (Category-3) sites stabilized at about one-third of the level at untreated sites (Figure 1). This difference was maintained throughout the remaining six years of monitoring. Whether this reduced abundance indicates that washed sites have yet to fully recover will be the subject of continued monitoring. With only three sites included in each of the categories and no before-impact data, however, there is no *a priori* reason to expect the infaunal communities at control and impacted sites to be similar. It is possible that even in the absence of impacts, infaunal differences of this magnitude could occur naturally among the small groups of widely separated sites. These differences can arise from zoogeographic variability or as described below, from inherent variability in the physical environment. Because differences in absolute abundance levels may be unrelated to impacts, traditional recovery assessment techniques (Ganning, 1984; Boesch *et al.*, 1987), which test for coincidence at control and impact sites, are likely to miss recovery detected by tests for parallelism.

During the period of stable abundance levels (1992-1997), the lower average infaunal abundance at washed sites (Category 3) was associated with a much different infaunal assemblage than that of control sites. This high β-diversity (see Appendix A.1) among categories was strongly related to variation in the fine sediment fraction among the monitoring sites. Specifically, the canonical correspondence between all post-recolonization (>1992) infaunal data and available physicochemistry, revealed a highly significant ($p \leq 0.001$) relationship between infaunal communities and the very fine (63-μ) sand fraction. All Category-3 sites were comparatively devoid of fines and supported a low-abundance infaunal community structure that was consistent with coarser sediments. Hence in 1992, the infaunal community at washed sites had achieved equilibrium

with ambient conditions and remained stable throughout the following years. Consequently, '*recovery*' as traditionally defined in terms of complete infaunal similarity between impacted and control populations, would not be expected to occur at the Category-3 sites under present conditions. It would only be expected after a change in the physical environment at Category-3 sites, namely, deposition of fine sediments.

Thus, the important remaining question for infauna is whether invasive cleanup and treatment processes were responsible for the removal of fine-grained sediments or whether the selection of the three washed sites with coarse-grained sediments was merely serendipitous. If they were devoid of fine sediments prior to the spill, then there would be no reason to expect their abundance levels to ever achieve that of the controls. If not, then recovery may not be quite complete insofar as a return to pre-spill conditions. In that case, full recovery will probably not occur at washed sites for many years, if at all. Long-term infaunal recovery then becomes a function of site-specific depositional rates. While the parallelism tests are capable of resolving the relatively rapid (intra-annual) recovery in infaunal abundance after 1991, they currently lack the statistical power to resolve very gradual changes in infaunal communities related to coastal processes. Each additional year of monitoring substantially increases the power to resolve these subtle trends in community properties.

Until this remaining question is resolved, the full impact of aggressive shoreline cleanup on infaunal communities cannot be determined. The analyses reported here indicate that impacted infaunal communities recovered abruptly over a one-year period following two years of low abundance resulting from oil-spill and cleanup impacts. The chronology of infaunal stabilization was about the same at both categories of impacted sites, with or without invasive shoreline cleanup. If the invasive cleanup techniques were responsible for eroding fine sediments from the lower intertidal zone, then infaunal impacts may last well beyond the eight-years of currently available monitoring data.

Because of this, the analyses reported here suggest that intertidal monitoring within Prince William Sound should continue, albeit with a slight redirection in effort. At a minimum, future surveys should include the nine lower intertidal sites analyzed in this report. Because of their temporal continuity, they act as sentinels for any subtle trends related to long-term recovery. Second, the strong relationship between grain-size and infauna points toward an expanded sampling program for physicochemical parameters. For example, variability in grain-size among infaunal replicates within sites could be examined to determine the degree to which small-scale variability

influences infauna. Currently available replicate grain-size samples from the 1995 survey are unsuitable because the finest fractions were imprecisely determined using volumetric techniques.

Finally, manipulative experiments could be conducted to further address the remaining question concerning the degree of ongoing infaunal recovery at Category-3 sites. These experiments could include hot-water washing of limited portions of nonoiled control areas. This would more directly assess the infaunal and physicochemical impacts from the invasive shoreline cleanup techniques similar to those applied in 1989. In addition, fine-grained sediment could be artificially introduced in portions of Category-3 sites to monitor infaunal repopulation and sediment dispersal. Various coastal process studies and modeling could also be conducted to determine whether the lower intertidal zone of Category-3 sites is naturally dispersive with respect to fine sediments.

Epibiotic Recovery

Recovery of intertidal epibiota, which live on top of rock surfaces, was assessed using the same methods that were applied to infauna. Originally, three tidal elevations for the epibiota were sampled during the annual surveys. However, the recovery assessment in this report includes only the upper and middle intertidal transects where epibiota were consistently measured on an annual basis through all surveys between 1989 and 1997. Three distinct epibiotic assemblages, representing different taxonomic groups, were examined. Percent algal cover, invertebrate cover, and invertebrate densities were analyzed separately. Invertebrate percent cover quantified sessile organisms such as barnacles and mussels whereas those based on counts included mobile invertebrate forms such as marine snails and limpets.

Of the three epibiotic assemblages, algal cover exhibited the most distinct pattern of recovery in the middle-intertidal zone. Although there were slight one-year differences in timing, the recovery signature observed for algal cover was similar to that seen for infauna. Algal cover in the middle intertidal zone increased significantly after 1990, a year earlier than the onset of infaunal recovery. By 1992, algal cover had achieved stable levels comparable to those at control sites. Almost all of the algal recovery was due to a rapid (intra-annual) increase in the coverage of *Fucus gardneri*. *Fucus gardneri* is a dichotomously branched brown algae (Phaeophyta) that is commonly referred to as rockweed or popweed. Other algal species were less abundant and were primarily understory species occurring beneath the *Fucus gardneri* canopy.

Algal cover in the upper intertidal zone was an order of magnitude smaller than in the middle intertidal. Nevertheless, the same recovery signature that was seen in the middle intertidal zone,

was also evident along upper intertidal transects. Recolonization occurred over a longer period at the upper transects where algal cover stabilized in 1993 as opposed to 1991. Protracted recolonization along upper transects may be a consequence of prolonged exposure to trace hydrocarbon contamination that was observed within the upper tidal reaches. Again, rockweed (*Fucus gardneri*) was the most dominant algal species found at the upper elevation followed by the red algae (Rhodophyta) *Endocladia muricata*, which is characterized by stiff, spine-covered branches. Compared to control sites, algal cover at both Category-2 and Category-3 sites showed the same recovery trend.

Although there was visual evidence of repopulation, statistically significant evidence of an initial impact and subsequent recovery of sessile invertebrates was not found at either Category-2 or Category-3 sites at either the upper or middle tidal elevations. These sessile filter feeders included the *Semibalanus* barnacle, and the large (*Balanus*) and small (*Chthamalus dalli*) acorn barnacles, as well as the mussel *Mytilus* cf. *trossulus*. Because these species reside in calcareous shells, their apparent lack of significant recovery may be due to the difficulty in distinguishing between live and dead organisms. This may have led to overestimates of live invertebrate cover during the early surveys immediately following the oil spill.

Despite the lack of statistical significance, an abrupt eight-fold increase in sessile invertebrate cover was visually evident at Category-3 (washed) sites between 1989 and 1991. In contrast, no similar evidence of impact and recovery of sessile invertebrates was apparent at Category-2 sites where less aggressive oil-spill cleanup procedures were applied. This suggests that hot-water washes were substantially more detrimental to sessile invertebrates than hydrocarbon contamination alone.

In contrast to invertebrate cover, the sharp increase in the abundance of motile invertebrate species at Category-3 (washed) sites between 1989 and 1991, was statistically significant. Total populations increased about twenty-fold within the middle-intertidal zone and nearly forty-fold along upper transects. Recovery was primarily due to recolonization by the checkered (*Littorina scutulata*) periwinkle along upper transects, while recovery of the Sitka (*Littorina sitkana*) periwinkle was more influential in the middle tidal reaches. This zonation can be ascribed to differences in the reproductive strategy of these two littorine snails. Limpets (Lottiidae), predatory snails (*Nucella*), and scavenger hermit crabs (*Pagurus*) also contributed to the recovery process, but had different recolonization rates.

Précis

Overall, similarity in the recovery process for a wide range of intertidal assemblages was noteworthy. Recovery began less than three years after the spill and required about one to two years before populations stabilized in 1993. The average eight-fold population increase was large enough to be detected by statistical tests. Subsequent fluctuations in population were smaller and could not be distinguished from those at control sites. The scope of this abrupt repopulation event provides compelling evidence that intertidal populations as a whole, had experienced a significant amount of recovery by 1993. Statistically significant recolonization was evident throughout the intertidal zone at both Category-2 (oiled) sites and Category-3 (hot-water washed) sites. Widely disparate intertidal assemblages, including infauna, algae, and epifaunal invertebrates participated in the recolonization.

Although a substantial portion of intertidal recovery had been completed by 1993, some important aspects remain unresolved. These warrant continued monitoring. For example, subtle trends within impacted populations remain visually evident in time profiles after 1993; but they cannot be rigorously quantified with parallelism tests applied to the current database. Future surveys will provide additional degrees-of-freedom for hypothesis testing and more-complex population models can be applied to describe the post-recolonization dynamics. In addition, average infaunal abundance at Category-3 sites stabilized at a level well below that of the control sites. Whether this is simply an artifact of nonrandomization in the study design or if impacted infauna have yet to return to pre-spill conditions, can be explored with continued monitoring and manipulative experiments.

Chapter 1. Introduction

In 1989, the *Exxon Valdez* accident spilled 11 million gallons of oil in Prince William Sound and outer coastal areas of the Gulf of Alaska. Approximately five million gallons impinged on 400 miles of shoreline and became stranded on intertidal habitats (Spies *et al.*, 1996). Oil coated rock surfaces, penetrated into soft sediments, and impacted a wide range of intertidal organisms. Cleaning removed a large amount of stranded oil but also damaged the intertidal environment. High-pressure hot-water washing was particularly destructive (Mearns, 1996). Based on a comparison of community parameters at unoiled, oiled, and washed sites, Driskel *et al.* (1996) and Houghton *et al.* (1996) concluded that by 1992, recovery of intertidal organisms at washed sites still lagged significantly behind the oiled sites. The longer term data and different statistical analysis techniques presented here, shed new light on these conclusions.

Background and Purpose

Shoreline-monitoring studies were implemented by the National Oceanic and Atmospheric Administration (NOAA) in Prince William Sound shortly after the *Exxon Valdez* oil spill. Multidisciplinary in nature, these studies focused on the geomorphology, chemistry, and biology of impacted sites to obtain information on biological recovery processes and the physical fate of oil in the intertidal zone. Results of geomorphology and chemistry studies are provided in several reports by Michel and Hayes (*e.g.*, 1994, 1998) and Roberts *et al.* (*e.g.*, 1997). Results of previous intertidal biological studies, which have been underway since 1989, are available in annual reports prepared by Houghton *et al.* (*e.g.*, 1996, 1997a, 1997b).

This report focuses on the timing and magnitude of the recovery of intertidal organisms based on data collected annually since 1989. It also includes more-recent data collected in the summer of 1997. Subsequent reports will emphasize other topics, such as the spatial and temporal variability of intertidal data and their implication for the design of oil-spill monitoring programs. Still other planned reports will investigate biological succession during recovery and biological impacts from changes in the physical environment that result from various oil-spill cleanup techniques.

The recovery process examined here refers to a specific biological response in a particular community, namely repopulation of intertidal organisms. In this context, '*recovery*' refers to a period when impacted intertidal populations stabilize and begin to track natural fluctuations observed at control sites. This is only one aspect of oil-spill recovery that includes socioeconomic recovery as well as recovery of the ecosystem as a whole. Even within intertidal communities, recovery can be measured in different ways. In controlled experiments, recovery can be defined as con-

vergence in community composition at oiled and unoiled sites. However, as described below, this approach is problematic when '*It is likely that oiled and non-oiled environments would systematically differ in ways other than oiling*' (Spies *et al.*, 1996).

Similarly, there are a variety of intertidal recovery issues related to population dynamics, including recruitment, trophic interaction, succession, and senescence (aging) of individual cohorts. Although this report examines some of the taxa that exhibited long-term recovery, the emphasis is on the broadest community measure; namely, the total abundance of major intertidal assemblages. Total abundance provided an unequivocal measure of the recovery process. The onset of recovery was clearly delimited by an abrupt one-to-two-year increase in total abundance at impacted sites. The increase was not observed in control populations. Comparing temporal fluctuations in abundance at control and impact sites provided a quantitative statistical method for identifying subsequent stabilization (recovery) of impacted intertidal communities.

Although this report finds that intertidal assemblages, as a whole, stabilized early in the monitoring program, there are compelling reasons to continue shoreline biological monitoring within Prince William Sound. For example, there is evidence that some individual species, such as the commercially and recreationally important little neck clam (*Protothaca staminea*), are still in the process of recovering (Shigenaka *et al.*, 1999). Future sampling years will supply additional degrees-of-freedom needed to resolve subtle temporal trends in the population of these individual taxa. Also, slight redesign of the field sampling program will lend needed insight into spatial and temporal covariability of intertidal biological and physicochemical data. Variance estimates have implications for the design of future oil-spill monitoring programs. Redesign of the sampling program will also allow further assessment of the impacts from habitat disturbance caused by aggressive shoreline cleanup techniques. Continued long-term monitoring will help place the magnitude of anthropogenic (oil spill) impacts in perspective with regard to natural fluctuations caused for example, by the El Niño–Southern Oscillation. Finally, from a purely scientific standpoint, continued monitoring will allow biological processes such as recruitment, succession, and trophic interaction to be investigated over the long term.

To highlight potential differences in the intertidal recovery process, this report is organized by taxonomic assemblage. First, the rationale for the analysis approach is described in the following sections of this chapter. This analysis approach differs from prior studies (Houghton *et al.*, 1996, 1997a, 1997b) in that recovery is evaluated using logarithmic transformations of abundance and percent cover. In addition, recovery is determined by abundance at impact and control sites becoming parallel over time rather than equal in magnitude. The second chapter describes the re-

covery of infaunal organisms that reside within intertidal sediments. Their abrupt repopulation was observed at impacted sites early in the monitoring program (*ca.* 1992). This abrupt recovery was followed by a sustained period of comparative stability in the infaunal population. Differences in the recovery process are described for various infaunal taxa. Chapter 3 focuses on the recovery of intertidal epibiota, which reside on the surface of hard substrata. They include algae, as well as sessile and motile invertebrates. Although the behavior, morphology, and tidal elevation of all of these intertidal assemblages differ greatly, the recovery process among them is remarkably similar. These similarities are the subject of the concluding Chapter 4 where a number of important questions about recovery are posed and discussed.

History

This intertidal monitoring program began in 1989. While the study's initial objective was to examine the effects of oiling and cleanup on intertidal epibiota and infaunal communities, in recent years, understanding and documenting their recovery processes have become of equal and perhaps of greater importance. The majority of sites presently sampled were selected in 1989 or 1990 based on their degree of oiling and their cleanup history. Sampling sites are primarily located in the western portion of Prince William Sound although one control site (Sheep Bay) is located to the east (Figure 2). Other unoiled control sites are located to the south (Hogg Bay and Bainbridge Bight) and in areas (Eshamy and Crab Bays) sheltered from the majority of spilled oil that spread within the Sound. Oiled sites, both washed and untreated, are located along the island chain extending from Naked Island in the North, around Knight Island, and to Latouche Island in the South.

Sampling sites were initially categorized according to treatment and oiling history as follows:

Category 1 – unoiled in 1989 with no cleaning treatments applied,

Category 2 – oiled in 1989 and untreated or cleaned only with cool-water flushes in 1989 (some bioremediation treatments may have been applied in 1989 through 1991),

Category 3 – oiled in 1989 and treated with hot-water, high-pressure washes one or more times, bioremediation applied at many sites in 1989 through 1991.

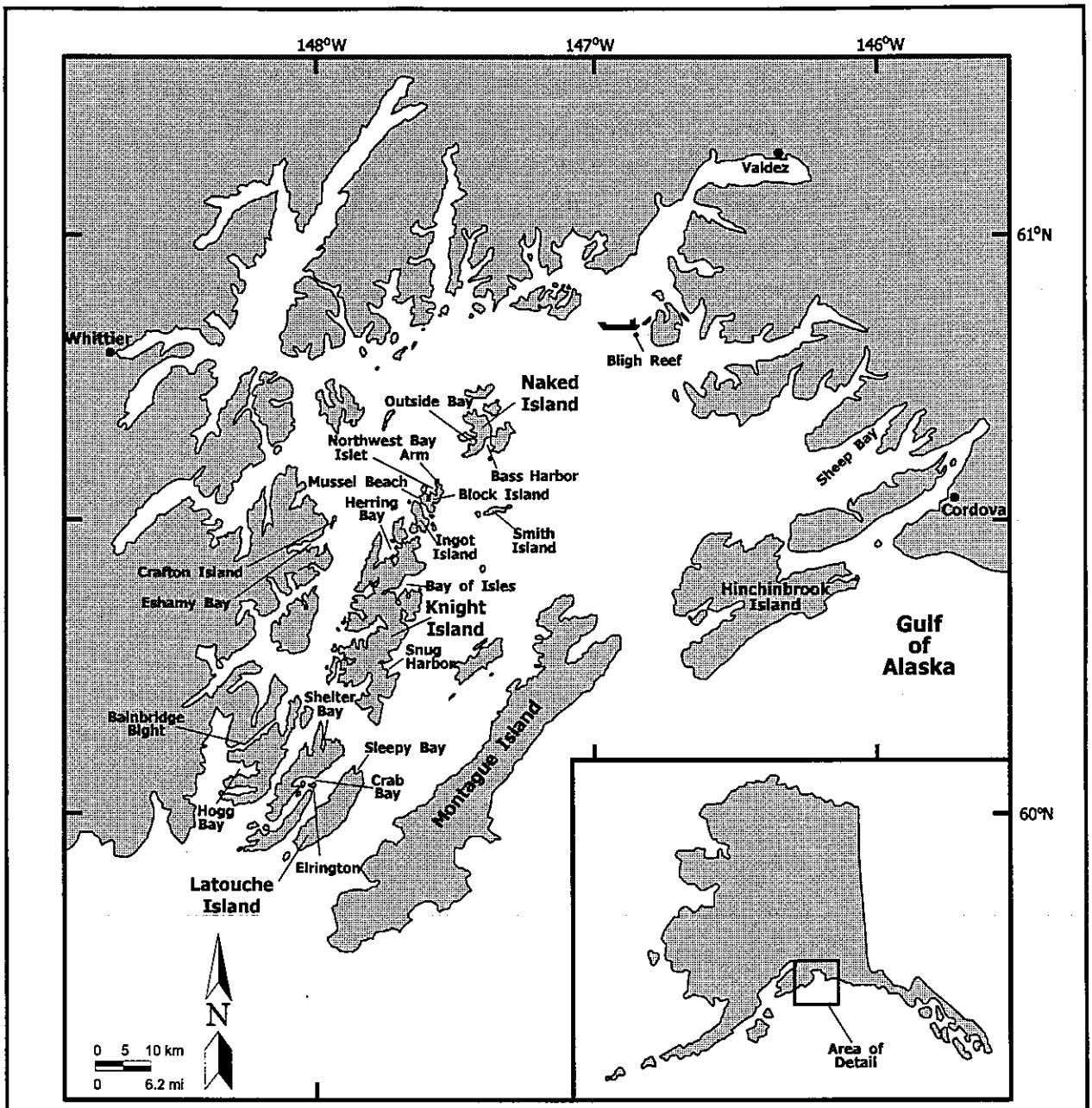


Figure 2. Intertidal sampling locations within Prince William Sound

The rationale for categorizing sites is provided in Appendix A-1 of Houghton *et al.* (1993a). Original treatment records and histories for the various locations were provided by Exxon, the State of Alaska, and personnel present at the sites in 1989.

At each sampling site, a stratified random design was initially used to select sampling locations along transects located at the upper, middle, and lower tidal levels. Transect lines were estab-

lished according to the biological communities present within each tidal level. The upper transect was established within the *Verrucaria* (lichen) zone above where rockweed (*Fucus gardneri*) is prevalent, the middle transect was located within the dominant rockweed (*Fucus gardneri*) zone, and the lower transect was positioned just below the *Fucus* zone near mean low water. Because the local intertidal biology dictated the cross-shore location of transects, the actual tidal elevation of the upper, middle, lower transects varied from site to site. Epibiota were sampled at five or ten replicate 0.25 m^2 quadrats at two to three tidal levels. While infauna were initially sampled at both the middle and lower levels, only the lower elevation has consistently been sampled and analyzed through the years.

Statistical Approach

Overview

In the years immediately following the *Exxon Valdez* oil spill, the primary goal of this monitoring program was to compare the amount of damage done to intertidal communities by oiling and shoreline cleanup methods. In recent years, the primary interest has been to determine if impacted intertidal communities have returned to pre-spill conditions. Because of this interest, the statistical approaches used in this report focus on the recovery of intertidal epibiota and infauna from the initial oiling in 1989 and the cleanup that followed.

As is inherent to most accident assessment studies, no pre-spill baseline information was available for the intertidal communities exposed to spilled oil. The only available information was post-spill data on organism abundance at 3 oiled, 3 oiled and washed, and 3 nonoiled sites in Prince William Sound. Hence, recovery cannot be simply based on the notion of a return to intertidal population levels prior to the spill because the pre-spill levels are unknown. Nor can assessment be based simply on a comparison between mean response levels at control and impacted sites because they may have been different prior to the spill. Instead, inferences concerning recovery must be based on indirect evidence that oil-exposed sites have returned to an unimpaired condition.

Temporal profiles of population response, namely changes in abundance at the control sites over time, depict how nonimpacted sites have behaved in Prince William Sound since the 1989 spill. These profiles, which track the average density of control populations, can be compared with temporal trends at impacted sites. At control sites, time profiles reflect localized conditions as well as population responses to regional changes in oceanographic and climatic conditions that affect all or most sites in the Sound. If each of these control sites respond to these regional cli-

matic changes over time similarly, then their temporal profiles should show similar trends over time as well. Actual organism density may differ among the control sites because of differences in local conditions, but the changes over time would be similar if these sites are responding to the same temporal changes in the regional environment.

Prior to statistical analysis, it is important to transform the data to achieve additive effects (Bartlett, 1947; Eberhardt, 1978). To attain additivity, this investigation of recovery analyzes temporal profiles of logarithmically transformed population data. As such, it departs from the approach previously used to assess intertidal recovery where data were displayed and analyzed on an arithmetic scale.

There are many reasons to compare populations on a logarithmic rather than arithmetic scale. In particular, the processes of natality and mortality tend to have multiplicative effects on population abundance and density. For example, favorable conditions can be expected to increase densities across sites by 20%, rather than adding 20 more individuals regardless of baseline abundance levels. Similarly, detrimental conditions are more likely to produce similar fractional reductions in abundance rather than equal reductions in absolute counts. These geometric changes have been a fundamental premise of population dynamics for two hundred years (Malthus, 1798). In addition, chronic effects of a toxicant (such as oil) are also likely to have a multiplicative (fractional) effect on populations. This is implicitly acknowledged by the use of '*Effective Concentration*' (EC) in toxicity tests. EC50 is the concentration of a toxicant that is calculated to affect 50% of the test population, not 50 individuals.

Time profiles of control populations within Prince William Sound demonstrate that regional influences tend to result in multiplicative changes in abundance rather than additive changes. This is evident from a comparison of mean species abundance plotted on the arithmetic (additive) scale and the same data plotted on a logarithmic (multiplicative) scale. Temporal trends in mean abundance are much more consistent among control sites when plotted on a logarithmic scale rather than on a linear scale.

For many species, logarithmic transformation of the data suggests that control populations are '*tracking*' or paralleling each other on a multiplicative scale. For example, Figure 3 illustrates time profiles for percent cover of the small acorn barnacle (*Chthamalus dalli*) for the three control sites. Profiles for mean percent cover (Figure 3a) suggest erratic changes in abundance over time across the three sites. However, after logarithmic transformation, the time profiles show differences in amplitude but also trends that are more-closely parallel over the nine years of sampling. The time profiles in Figure 4 show a similar effect from logarithmic transformation of the

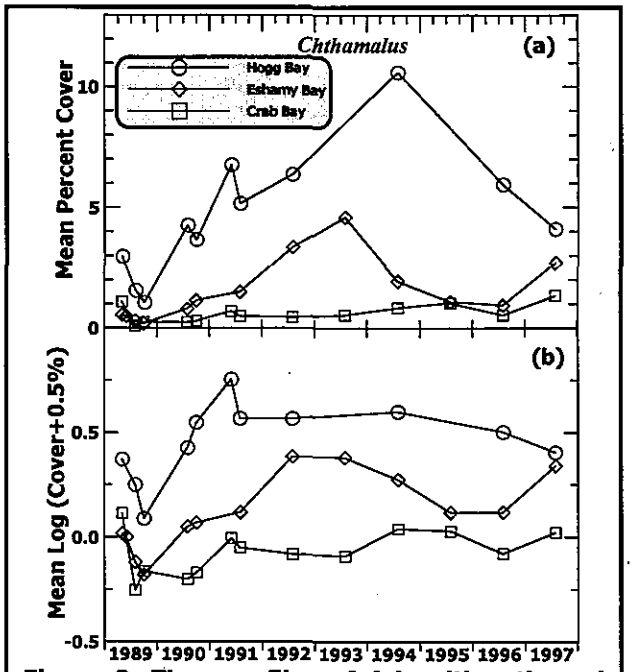


Figure 3. Time profiles of (a) arithmetic and (b) logarithmic mean percent cover of the small acorn barnacle (*Chthamalus dalli*) at control sites

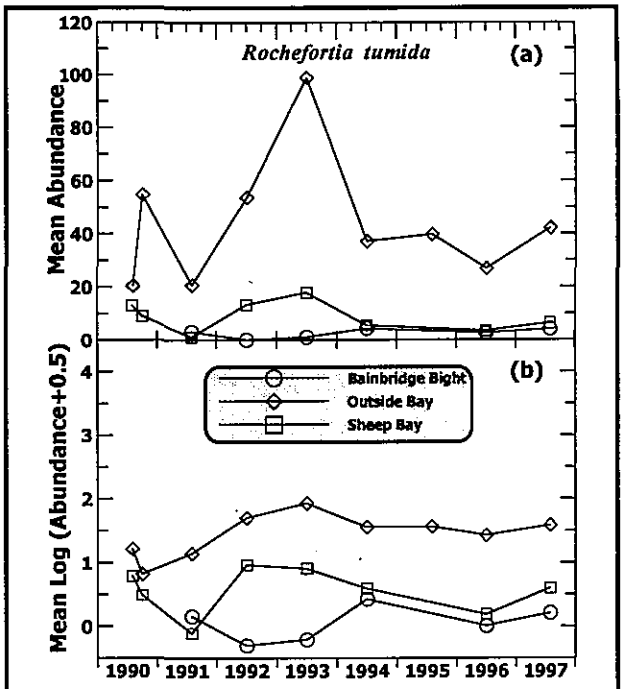


Figure 4. Time profiles of (a) arithmetic and (b) logarithmic mean abundance of the bivalve *Rochefortia tumida* at control sites

abundance of a small (< 4 mm) infaunal bivalve (*Rochefortia tumida*). Averaging the logarithmic transform of replicate sample abundance is comparable to computation of a geometric mean. It reduces the influence of outliers and is a common technique for minimizing the correlation between the mean and variance in multiplicative distributions.

The words of Sokal and Rohlf (1997) provide some solace for the use of transformations in general, and for logarithmic transforms in particular:

> At this point many of you will feel more or less uncomfortable about what we have done. Transformation seems to (*sic*) much like 'data grinding.' When you learn that often a statistical test may be made significant after transformation of a set of data, though it would not have been so without such a transformation, you may feel even more suspicious. What is the justification for transforming the data? It takes some getting used to the idea, but there is really no scientific necessity to employ the common linear or arithmetic scale to which we are accustomed. Teaching the 'new math' in elementary schools has done much to dispel the naïve notion that the decimal system of numbers is the only 'natural' one. It takes extensive experience in science and in the handling of statistical data to appreciate the fact that the linear scale, so familiar to all of us from our earliest experience, occupies a similar position with relation to other scales of measurement as does the decimal system with respect to the binary or octal numbering systems and others. If a relation is multiplicative on a linear scale, it may make much more sense to think of it as an additive system on a logarithmic scale.

Under the hypothesis of an oil spill effect followed by eventual recovery, control and impacted profiles would diverge upon impact, and over time, begin to 'track' or parallel each other as impacted sites begin responding solely to the same regional climatic changes or oceanographic conditions as the reference or control sites. Hence, eventual parallelism between mean profiles for control and oil-treatment sites would provide strong inferential evidence of recovery (Skalski, 1995). To see this recovery, the data must be analyzed on the proper scale. 'Tracking' or parallelism of the control sites after logarithmic transformation of the data strongly suggests that the proper scale to utilize for assessing recovery is the logarithmic scale.

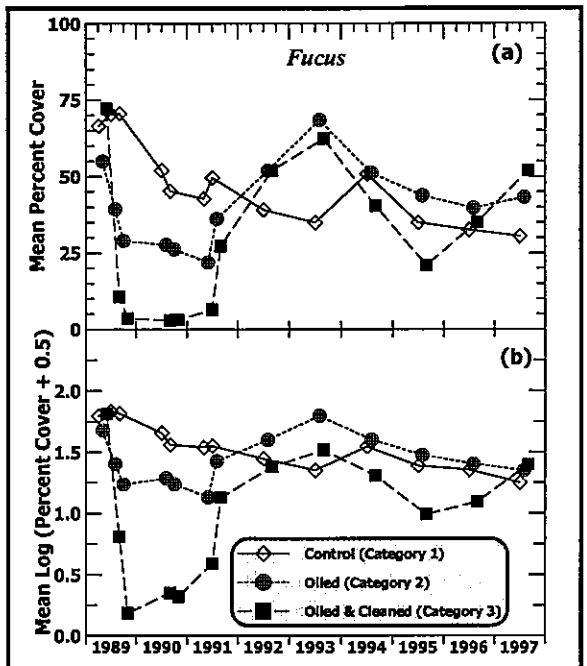


Figure 5. Time profiles of percent cover of rockweed (*Fucus gardneri*) at control, oiled, and oiled-washed sites for (a) untransformed and (b) logarithmically transformed data

Another example illustrates the advantages of logarithmic transformation in recovery assessments. In Figure 5a, untransformed mean profiles for control, oiled-washed, and oiled and untreated sites are plotted for percent rockweed (*Fucus gardneri*) cover through time. These profiles show large fluctuations in cover that differ among the treatment categories throughout the monitoring program. On the other hand, the mean computed from logarithmically transformed cover (Figure 5b) clearly illustrates an initial divergence of the oiled sites from the control profile. This initial divergence is followed by an extended period when both categories of impacted sites closely parallel the control profiles. Thus, on the logarithmic scale, recovery of rockweed (*Fucus gardneri*) is more apparent and persistent.

The purpose of the statistical analysis utilized in this report is to objectively assess whether impacted sites have recovered in Prince William Sound. From one perspective, recovery can be considered complete when impacted intertidal populations eventually begin to track or parallel the control site profiles. Under that scenario, the statistical tests of recovery are equivalent to a test for parallelism. Moreover, at any given time, differences in population levels between control and impacted sites are not a consideration; only the relative temporal trends are of interest.

This is an important aspect of the recovery analysis because there is no *a priori* reason to expect mean population levels at control and impacted sites to be the same, even in the absence of the spill. On the contrary, arbitrary differences in average populations are likely to occur because there are only three sites included in the computation of the mean within each treatment category, and because large population differences occur naturally among the widely separated sites. These natural population differences are evident in the temporal profiles at control sites shown in Figure 3 and Figure 4. For example, the mean cover of the small acorn barnacle (*Chthamalus dalli*) is over three times higher at the Hogg Bay site than at Crab Bay. In each of the eleven surveys conducted at these sites to date, this difference was statistically significant ($p \leq 0.10$).

This measurable natural difference in the populations has profound implications for the interpretation of recovery based solely on absolute differences between control and impact sites. Suppose a different set of control sites had been selected with environments that all happened to be similar to Hogg Bay. Then, the mean control population of small acorn barnacles would be artificially elevated compared to randomly selected sites within Prince William Sound. Accordingly, mean populations at impacted sites may never reach these artificially elevated control levels, because the impacted sites may have never been as amenable to small acorn barnacles to begin with. It would be incorrect to assume that recovery had not occurred at impacted sites simply because they did not achieve the same absolute population level of the control sites.

In fact, systematic environmental differences between control and impact sites are expected for reasons unrelated to oiling or shoreline cleanup techniques. These arise from natural processes, such as the prevailing current flow as suggested by Spies *et al.* (1996). Because these differences cannot be randomized, recovery assessments based on a direct comparison of mean population levels at control and impact sites is statistically untenable. In contrast, tests based on differences in population trends over time, without regard to absolute population levels within a given year, provide a more viable method for assessing recovery. For that reason, error bars, which show within-year within-category variance and emphasize differences in absolute abundance, are not provided on the figures within the body of this report. The precision of multi-year trends is much greater than the precision that would be conveyed by error bars with only two or three degrees-of-freedom within a given year.

Nevertheless, insight into the dispersion of the computed averages may be of some interest for reasons unrelated to recovery assessment. For example, they could be used to gauge the spatial scale of population variability. This can be accomplished through a comparison of replicate variability at a particular site to variability among sites within a particular treatment category. Al-

though these issues are ancillary to the recovery assessment performed in this report, range limits and untransformed absolute abundance levels are shown in Appendix A.4 for the time profiles provided in the body of this report.

Statistical Tests

Returning to the same control and treatment sites year after year constitutes a multivariate repeated measures study (Morrison, 1976). Here, the same '*experimental*' units or sites are repeatedly measured on an annual basis, and as such, observations are correlated through time and cannot be considered independent. Repeated measures have often been mistakenly treated as independent replicate observations in ecological studies (Hurlbert, 1984). However, these data violate the assumption of independence necessary for most classical univariate statistical methods including regression, analysis of variance (ANOVA), and nonparametric statistical constructs. Hence, Skalski and Robson (1992) recommend using multivariate repeated measures and profile analysis to test for impact and recovery following an environmental accident. These methods properly account for the dependencies within the monitoring data. Unfortunately, for multivariate ANOVA to be effective, more replicate sites are needed than repeated measures over time. This is not the case with the Prince William Sound monitoring data where typically 3 replicate sites are collected during 9 or more resampling occasions per treatment category. The consequence is that *ad hoc* statistical analyses must be utilized and *p*-values associated with tests of significance should be only loosely interpreted when inferring impact or recovery. Otherwise, strict adherence to reported *p*-values from statistical tests might result in excessive Type I or II errors that lead to false conclusions or exaggerated confidence in the results.

In this analysis of Prince William Sound data, sequential testing procedures were used to test for recovery and to identify the periods of impact and recovery. Starting with the most recent years of data, a 6-year window of time was examined. Within this window, a test of parallelism (*i.e.*, recovery) was performed. If the null hypothesis of parallelism was not rejected, then the window was moved back one year in time and the analysis repeated. This back-step sequential procedure was continued until the null hypothesis of parallelism was rejected indicating that the period of impact had been detected. A forward selection procedure could also be performed, beginning with the initial period of immediate oil-spill impacts. The time window would be advanced by one-year increments until evidence of recovery is first detected (*i.e.*, null hypothesis of parallelism is no longer rejected).

The time span of the test window was fixed at 6-years so that parametric tests would have at least 4 degrees-of-freedom for the error term. Smaller time windows are more sensitive to localized

deviations from parallelism but have fewer degrees of freedom and much lower statistical confidence. Longer time windows have more degrees of freedom for testing but have inherently less temporal resolution to detect rapid (intra-annual) recovery events. Any choice of window size for the analysis is a compromise between these opposing forces of resolution and reliability.

Two closely related parametric models can be used to test for parallelism within a selected time window. The first model, based on a time-by-treatment ANOVA, is more involved but allows all three treatments (control, oiled, and hydraulically washed categories) to be tested simultaneously. The second model, involving logarithms of ratios of mean values, is easier to interpret and apply. The second, logarithmic-ratio approach was used in all the tests whose results are described throughout this report. However, the two approaches were found to give similar results in several confirmatory analyses and the first approach is described here for its didactic value.

The first approach fits a common polynomial to the mean values over time and tests whether control (C) and treatment (Category-2 or 3) sites (T) share the same temporal trends. A stepwise regression procedure is used to identify the optimal polynomial form. At each step, unique intercepts for the different treatments are modeled and parallelism is reflected by the significance of regression coefficients for temporal terms that apply to specific treatments or controls. If a model with a single set of temporal terms, common to both treatments and controls, adequately describes the trends, then the profiles are considered parallel.

For example, consider the case with a single treatment (T) and control (C), and where there is a quadratic relationship with time. The ANOVA would be of the form shown in Table 1 where n is the number of years included in the test.

Table 1. Time-by-Treatment ANOVA test for parallelism

Source	degrees of freedom	Description	
Total	$2n$		
$Total_{cor}$	$2n$-1	Common intercept	
Treatment	1	Unique intercepts	
T	1	Common linear component of time	
T^2	1	Common quadratic component of time	
Treatment x T	1	Unique linear trends	} F-test 2, $2n$-6
Treatment x T^2	1	Unique quadratic trends	} Test of parallelism
Error	$2n$-6		

Here, the test for parallelism and recovery is equivalent to a test for a time-by-treatment interaction. Separate intercepts are fit because there is no *a priori* reason to assume the control and impacted sites would recover to a common mean value. This is because treatments can be con-

founded with location differences and because there is a lack of randomization for assuring equality in the expected means.

The second closely-related approach, which is used throughout this report, is based on the difference in the logarithm of means for each year. This difference can be cast as the logarithm of the ratio of the means *i.e.*,

$$\log\left(\bar{x}_C\right) - \log\left(\bar{x}_T\right) = \log\left(\frac{\bar{x}_C}{\bar{x}_T}\right) \qquad \textbf{Equation 1}$$

As a consequence, this approach has the advantage that the successive logarithmic ratios are largely uncorrelated, allowing standard statistical methods of inference. Regression of a single polynomial on the logarithmic ratios provides a test for the null hypothesis of parallelism, which is rejected if the regression coefficients are significantly different from zero.

In addition, Equation 1 makes it clear that a test for parallelism is based on changes in the relative amplitude (ratio) of population fluctuations in addition to their temporal concurrence. Thus, parallelism tests differ fundamentally from tests of synchrony, where only temporal coincidence is required regardless of amplitude. They also differ from tests for convergence, where absolute population levels at control and impacted sites are compared through time. Taken separately, neither synchrony nor convergence criteria are suitable for statistical confirmation of recovery. In contrast, a combined statistical construct, which is based on parallelism tests of logarithmically transformed abundance, provides a definitive means for quantifying intertidal recovery within Prince William Sound. In the following chapters, it is applied to a wide range of intertidal assemblages, taxa, and tidal elevations.

Chapter 1. Intertidal Infauna

Introduction

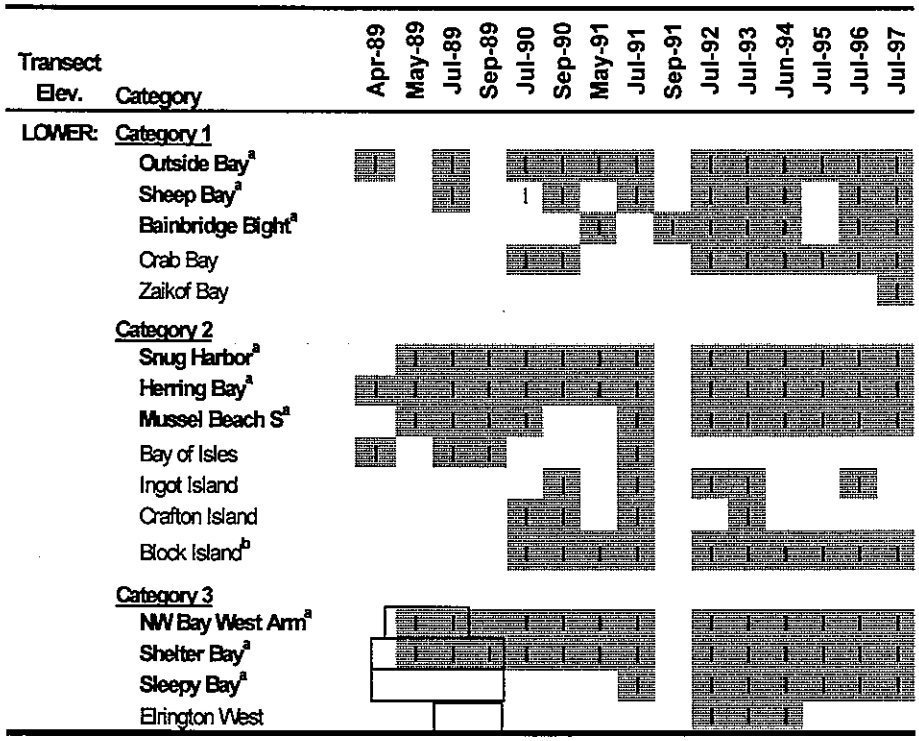
Mixed-gravel, pebble, and sandy-silt substrata are prevalent in the sheltered littoral environments of Prince William Sound. Within this granular environment, a distinct assemblage of marine invertebrates can be found. These sediment dwellers, or infauna, are well adapted to interstitial living and mostly consist of polychaetes (segmented worms), bivalve mollusks, and amphipod and decapod crustaceans. In contrast, epibiota, whose recovery is the subject of the following chapter, live on or attached to hard substrata. Epibiota encompass a broader range of marine organisms and include marine algae (seaweed), porifera (sponges), cnidaria (sea anemones), bryozoa (colonial moss animals), echniodermata (sea stars, sea urchins, and sea cucumbers), brachiopoda (lampshells), and sundry other phyla.

Infaunal invertebrates are economically and ecologically important because of their low trophic level within the marine food chain. They provide a food source for the more-mobile epifaunal and pelagic marine organisms such as crabs, fin fish, and marine mammals. Some species, such as littleneck (*Protothaca staminea*) and butter (*Saxidomus giganteus*) clams, are harvested for human consumption. Also, changes in the infaunal community provide an important indication of oil-spill impacts. Infauna have limited mobility and cannot easily escape exposure to contaminants in their environment. Also, because they reside within seafloor sediments, they are close to buried hydrocarbons that have persisted long after the spill (Hayes and Michel, 1998). Many infauna are filter feeders and bioaccumulate contaminants even when standard chemical assays are no longer able to detect pollution. In addition, these sediment dwellers provide insight into impacts from various shoreline cleanup techniques because they are especially sensitive to mechanical disturbance and to lasting changes in grain-size distributions.

Spilled oil and the subsequent cleanup effort can affect infaunal organisms by directly killing them or by affecting behavior that indirectly results in mortalities. There is a rich literature on the effects of oil on infauna and Fukuyama and VanBlaricom (1997) and Fukuyama *et al.* (1998) have summarized studies for subarctic infaunal communities. A factor that often affects recovery of infaunal communities is retention of oil within sediments (Clark, 1982). Although oil near the surface weathers, anoxic conditions deeper in the sediments slows weathering and infauna may continue to be exposed to trapped oil for many years.

Using monitoring data through 1992, Driskell *et al.* (1996) concluded that infaunal recovery had begun within Prince William Sound. However, several community indices including total abundance, species richness, and diversity, as well as abundance of major infaunal taxa lagged at oiled and washed (Category-3) sites. Using additional data through 1996, Houghton *et al.* (1997b) suggested that infaunal recovery at oiled and washed (Category-3) beaches is progressing but not complete. To some extent, their assessment was based on observed differences between the infaunal community parameters at impacted and control sites. This report applied a different criterion for recovery based on temporal parallelism in population levels rather than convergence in community properties. It was also applied to a larger infaunal database that now spans nine years (Table 2).

Table 2. Infaunal samples collected from 1989 through 1997



Approximate time windows in which treatments occurred in 1989.

[a] Core group of stations used in calculation of category means.

[b] Category uncertain due to unknown cleanup history.

Methods

Field Sampling and Laboratory Procedures

Infauna were collected with a modified cylindrical clam corer measuring 10.5 cm in diameter by 15 cm in length. The area sampled was 0.009 m^2. Five replicate samples were collected at each of the 12 low intertidal sites during the July 1997 survey (Table 2). A separate sample was collected for grain size, total organic carbon (TOC), and total Kjeldahl nitrogen (TKN) analyses. Infaunal samples were collected at permanent rebar markers located along transect lines. Coring locations were offset from the markers by about 1 m to avoid resampling the same location. The infaunal abundance data are summarized in the Appendix B.1 and the physicochemistry in Appendix B.2.

Samples were live-sieved in the field using a 1.0-mm mesh sieve. All material retained on the sieve was placed into containers and preserved with a 10-percent solution of buffered formalin. Once in the laboratory, the samples were rinsed on a 0.5-mm mesh sieve and transferred to 70-percent ethanol containing Rose Bengal. Samples were sorted under a binocular dissecting-microscope. All organisms were separated into major phyla for subsequent identification by taxonomists expert in identifying organisms within specific taxonomic groups.

Infaunal Database Reassessment

Prior to analysis, the entire multi-year infaunal database was extensively reviewed. A number of inconsistencies were found and corrected. Beyond mismatches between the database and the taxonomic key, species were identified to different taxonomic levels or assigned different but nearly synonymous names depending on the year. Unless corrected, this variability in taxonomic identification introduces erroneous interannual variability that affects the ability of statistical tests to discern impacts and recovery. Most of the inconsistencies and inaccuracies were corrected through consultation with taxonomists. However, in the case of mollusks, the original samples were reanalyzed. One of the errors in the original mollusk database was enumeration of organisms that were dead prior to collection as indicated by the presence of empty shells. Exclusion of these organisms is crucial to impact assessment studies where the impacts themselves may have caused the mortality.

In addition, a large number of identified taxa were determined to be epifaunal or meiofaunal, and were excluded from the analysis. Epifauna do not reside within granular interstices and meiofauna are smaller than the sieve size (1.0 mm). For example, the arctic saxicave bivalve (*Hiatella arctica*) was considered an epifaunal species in this reassessment because it is a nestling species

that lives along the cracks and crevices on the surface of rocks (Morris *et al.*, 1980). Of the 79,000 specimens included in the original database, only about 35,000 or 44% were positively identified as infauna by taxonomic specialists. The abundance data for the remaining epifaunal organisms were not combined into the epibiota database because of differences in sampling techniques and locations.

A subset of 22,500 infaunal organisms representing 121 individual species was used for multivariate community analysis and determination of diversity indices. By definition, diversity indices require an accurate assessment of the number of distinct species and their associated abundance. Therefore, it is inappropriate to include taxa that are not identified to species level and that may represent more than one distinct species (*i.e.,* multispecific taxa). Similarly, the primary purpose of multivariate ecological analysis is to identify trends in community structure that are related to environmental variation. Performing the analysis on multispecific taxa can mask important trends because individual species, even within the same genus, can respond differently to environmental influences. Thus, a multivariate analysis of multispecific taxa reduces ecological differentiation. In contrast, parallelism tests for temporal changes are based on total abundance, which is a measure of the entire community without regard to species structure. As such, time profiles of abundance are determined from the entire set of infaunal organisms regardless of their multispecificity. Consequently, abundance was computed using the entire database of strictly infaunal organisms. However, for reporting purposes, the 1997 database listed in Appendix B.1 includes all the enumerated organisms, irrespective of their designation as infauna, epifauna, or meiofauna.

Because of the changes made to the historical database, differences between the conclusions reported here and those of prior studies cannot be solely ascribed to the addition of another year of samples from 1997. The infaunal community in samples collected during 1997 was similar to samples collected in prior years. Instead, differences in the results reported here are due to changes made to the entire multi-year database and the application of a different statistical technique that focuses on assessing parallelism in global community parameters (*viz.*, total abundance).

Abundance Computations

To assess impacts and recovery from oiling and cleanup, three core sites were used from each of the three treatment categories (Table 2). Although infaunal sampling occurred in 1989, these data were excluded from the database because Exxon possesses the type specimens for that period of the study (Houghton *et al.*, 1997b). Data from samples collected in Crab Bay in 1990 were also

absent from the database. The remaining data encompassed the nine core sites and covered most of the eight years of infaunal monitoring conducted to date. Also, the cleanup and oil-exposure history was well documented for these nine sites. Because the cleanup treatment within the lower intertidal zone at Block Island was uncertain, it was excluded from the analysis even though it was also continuously sampled throughout the monitoring program.

Multiple surveys were conducted in 1990 and 1991 but only the July data were used to maintain seasonal consistency. The only exception was Bainbridge Bight where September 1991 data were used because a July survey was not conducted in 1991. This substitution does not appear to materially affect the composite mean despite the seasonal variability that is known to exist within infaunal communities (Figure 6). It is preferable to a substitution using May 1991 when early spring recruitment can introduce a stronger seasonal influence.

In general, the eight years of infaunal abundance data from nine sites provided a balanced design for comparing temporal trends among the three treatment categories. Nevertheless, one or two years of infaunal data were missing at three of the sites. To avoid temporal bias, the missing values were calculated using iterative procedures described in Section 15.3 of Snedecor and Cochran (1989). As described in Appendix A.2, this was necessary to reduce potential bias in category means that could arise from the absence of one

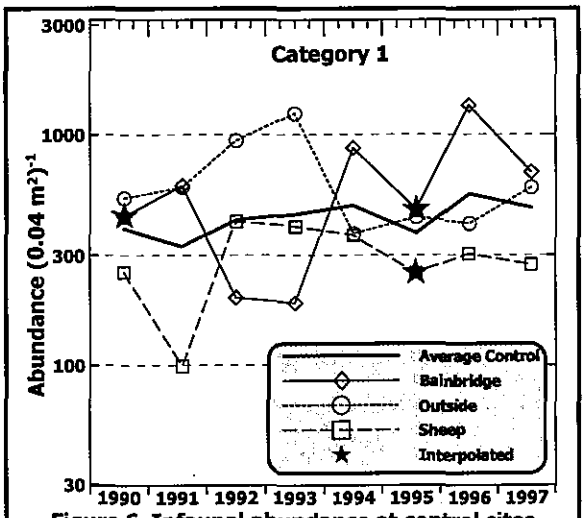


Figure 6. Infaunal abundance at control sites

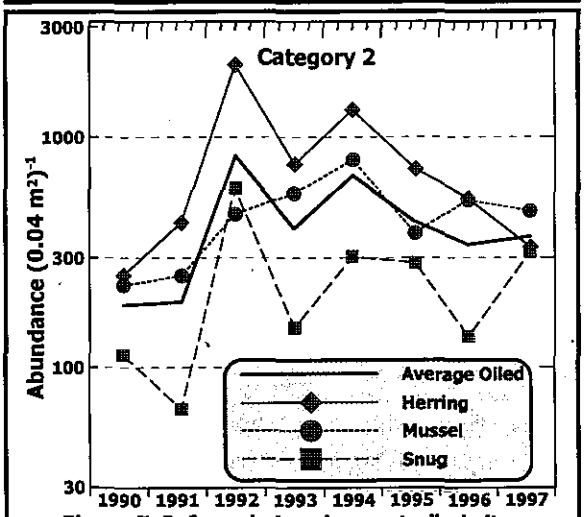


Figure 7. Infaunal abundance at oiled sites

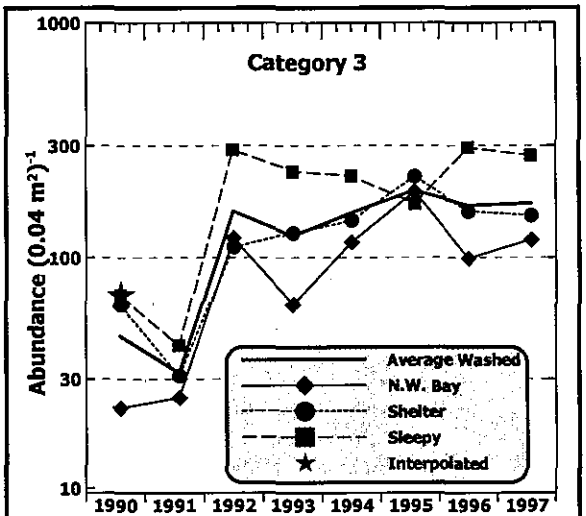


Figure 8. Infaunal abundance at washed sites

year of data at a site that is normally higher or lower than the other sites within that category. In reality, the ranking of site abundance used to compute category means varied from year to year as shown in the time histories of Figures 6, 7, and 8. Consequently, the four calculated points were in close proximity to the annual averages and the estimation process did not materially affect computation of category means or tests for parallelism.

Parallelism Test

The statistical tests for infaunal recovery evaluated the degree of parallelism between time profiles of abundance at control and impacted sites. The null hypothesis, that the time profiles were parallel, was tested through temporal regression on the logarithm of the ratio in mean abundance as formulated in Equation 1. Specifically, least-squares regression was performed on the mean abundance within a six-year period. The equation was of the form,

$$\log\left(\frac{\bar{x}_C}{\bar{x}_I}\right) = (m \times \text{time}) + b \qquad \textbf{Equation 2}$$

where m and b were the slope and intercept coefficients estimated in the regression, and $\bar{x}_C$ and $\bar{x}_I$ were the mean abundance at control (Category-1) and impact (Category-2 or 3) sites. If the regression generated a slope coefficient that was statistically significant (different from zero), then the time profiles within that six-year window were not considered parallel and recovery had not yet been achieved. Repeating the regression on six-year windows that start progressively later by one year often eventually yielded a slope coefficient that was not significant indicating that recovery has occurred. Statistical tests based on Equation 2 yielded results similar to the time-by-treatment ANOVA outlined in Table 1.

Terms in the regression equation that were quadratic in time (or of higher order) were not significant and Equation 2 was found optimal. Consequently, the fit within each 6-year time window was based on 4 degrees-of-freedom for the error term (6 samples minus one degree for the intercept and one for the slope). Regression using moving windows of a shorter (5-year) duration would only have 3 degrees-of-freedom and have very little statistical power to detect departures from parallelism. Conversely, longer time spans (>7 years) mask the large repopulation events that are found to occur over periods of only one to two years. Thus, regressions performed on a series of moving six-year windows balance the opposing forces of temporal resolution and statistical reliability.

The exponential population model incorporated in Equation 2 dates back 200 years (Malthus, 1798). It assumes no age structure, no seasonality, and unlimited resources yet, in spite of its simplicity, provides a robust representation of important features of population dynamics. It states that the average abundance $\bar{x}(t)$ of an assemblage of organisms at any given time t can be determined from an initial abundance $\bar{x}(t_0)$ as follows:

$$\bar{x}(t) = \bar{x}(t_0)e^{rt} \qquad \textbf{Equation 3}$$

where r is the population growth rate representing the difference between natality and mortality. Thus, in the comparison between two populations modeled by Equation 2, the slope coefficient (m) represents the difference in population growth rates and a significant departure from parallelism implies that the rates are measurably different at control and impact sites.

In theory, the statistical test for parallelism could be performed on any univariate infaunal community parameter, not just mean abundance. However, this report focuses on the most general measure of community health, namely total abundance. In large part, this is because of the limitations associated with other univariate measures such as the Shannon (Shannon and Weaver, 1949), evenness (Pielou, 1977), and dominance (Wittaker, 1965) indices. These diversity indices are unsuitable for describing potential impacts to infaunal communities because they lack biological meaning, show little correlation with environmental quality, are difficult to interpret ecologically, and often result in ambiguous or biased estimates of diversity (Goodman, 1975; Washington, 1984; Green, 1979). As a result, many of these univariate measures are not recommended for inclusion in monitoring programs (USEPA, 1987).

Another approach for examining recovery-related changes in community structure is through the multivariate analyses described below. Although this report only uses multivariate analyses to identify those taxa responsible for the marked repopulation event, they can be extended in future reports to elucidate biological succession related to the recovery process.

Multivariate Analysis

The parallelism test described above was applied to time profiles of total infaunal abundance and the abundance of major infaunal taxonomic groups. In many cases, the marked repopulation event at impacted sites between 1991 and 1992 resulted in statistically significant departures from parallelism. As described in the Results Section below, this indicates that these infaunal assemblages were measurably impacted and that their recovery was in progress during a period that includes 1991. The question then becomes: What specific infaunal taxa participate most in

the repopulation event and do their time profiles exhibit significant departures from parallelism? This can be answered in a quantitative manner using multivariate analyses. When environmental factors such as grain size are included in the multivariate analyses, the influence of natural conditions, unrelated to hydrocarbon exposure or shoreline cleanup, is revealed. This section presents a brief description of the multivariate analyses applied in this report.

Multivariate analyses, as defined in Appendix A.1, encompass a broad range of techniques. Although many yield similar results, careful implementation can significantly improve their utility. For example, their direct application to the entire infaunal database confounds spatial and temporal variability, and the effects of oiling and shoreline cleanup. This makes the resulting ordination diagrams particularly complex and difficult to interpret. In contrast, this report applied two separate sets of multivariate analyses to limited portions of the infaunal database to address two specific questions:

(1) Which taxa contributed to the recovery at impacted sites?

(2) How are infaunal differences among sites related to site-specific environmental properties?

First, to identify the individual taxa that were largely responsible for repopulation, a set of multivariate analyses were conducted on data from impacted sites (Category 2 and 3) only. This set of analyses would be diluted by the inclusion of data from control (Category-1) sites, where recolonization is not plainly evident. Second, to investigate the influence of natural variation in environmental properties, all data collected since 1992, after recolonization had occurred, were included in separate set of multivariate analyses. This avoided confounding natural geographic differences with strong temporal variability contributed by the recovery process.

In addition, different ordination techniques were selected to improve the efficacy of the two sets of multivariate analyses. The relative advantages of principal-component and correspondence analysis are described in Appendix A.1. Principal component analyses identified those taxa that materially participated in recolonization of impacted sites. Category-2 and 3 data were analyzed separately to emphasize temporal changes rather than zoogeographic or treatment-related variability. Within each analysis, the infaunal variability did not introduce strong nonlinearity, which would be evident as appreciable curvature or a horseshoe effect (see Appendix A.1) in the ordination diagrams. The principal component analyses used here employed a measure of infaunal community structure called CNESS (Appendix A.3). It accurately quantified infaunal differences among samples and balanced the influence of rare versus dominant taxa through selection of an intermediate subsample size (Grassle and Smith, 1976).

In contrast, the linear principal-component technique was not suitable for evaluating the influence of physicochemical properties on infauna because that database included samples from widely different locations having disparate communities (high β-diversity, see Appendix A.1). However, canonical correspondence analysis were capable of accommodating these unimodal species distributions (ter Braak and Verdonschot, 1995). Additionally, Monte Carlo permutation tests (Manly, 1991) were used to evaluate the significance of the relationship between infauna and physicochemical parameters. Over 1000 random permutations were used to resolve *p*-levels as low as 0.001. Also, prior to analysis, the environmental variables were transformed to improve the normality of their distributions and the homogeneity of variance. For total Kjeldahl nitrogen (TKN), a $\log_{10}(x)$ transformation was found to significantly improve the distribution. The variance of proportional variables, such as the grain-size fraction and TOC, was stabilized with an arc-sine transformation of the form $2 \sin^{-1}(\sqrt{x})$ (Draper and Smith, 1981). Homogeneity of variance is an important precondition for many statistical tests.

Grain-Size Analysis

Accurate determination of grain size can be important for interpreting the infaunal variability among samples. Strong relationships between infaunal assemblages and grain size have been well documented although it is unlikely that grain size alone determines the species distributions (Snelgrove and Butman, 1994). As will be shown in the following section, grain-size is significantly correlated with infaunal distribution within Prince William Sound. This is demonstrated through a canonical correspondence analysis (Appendix A.1) which is designed to elucidate any association between the distribution of intertidal infauna and environmental parameters in general. The statistical significance of these associations were determined directly through Monte Carlo randomization tests as described in the previous section and in Appendix A.1. Because grain-size was found to have the greatest influence on infauna and because a different methodology for determining grain-size was applied to the 1997 data, it is discussed in more detail here. Analysis methods for TOC and TKN in 1997 were comparable to previous years.

Only intermediate grain sizes, between very fine sands (>62.5 μ) and gravel (<4 mm), were included in the correspondence analysis. This size range encompasses the ambit of most macroinfauna. Larger size fractions were excluded because pebbles and cobbles, whose dimensions exceed a few percent of the 10.5-cm core, were probably undersampled. Smaller mud fractions (silt and clay) were excluded because they were not accurately determined by the wet-sieve/volumetric method applied prior to 1997. During that time, grain size was determined from the volume of water displaced by wet-sieved fractions (Houghton *et al.*, 1997b based on McNeil

and Ahnell, 1964). Although wet sieving is less time consuming than dry method, and can be accomplished in the field, volume measurements are inherently less precise than weight determinations. Also, prior to 1997, the smallest fractions were estimated from the particulate volume that accumulated in a settling-tube over a fixed time. Settling tube analyses are sensitive to sediment shape and packing (USACOE, 1984). Standard methods currently in use for grain-size analysis are based on fractional weight rather than displaced volume, and the finest fractions are accurately determined by weighing pipette extractions from sediment suspensions over time (Plumb, 1981).

Deficiencies in the volumetric method appear as bimodal grain-size distributions that were reported prior to 1997 (Table A-8 of Houghton *et al.*, 1997b). On average, both methods determined the primary mode, or median size, to be larger than gravel (Figure 9). Steadily decreasing size fractions through gravel and sand classes were also comparable. However, the volumetric method often found another mode in the grain-size distribution within the mud fraction. This is reflected in the statistically significant increase in the average mud fraction relative to very fine sands shown in Figure 9. The analyses conducted in 1997 were based on more-precise pipette determinations and found unimodal size distributions without a second mode in the mud fraction. Bimodal grain-size distributions are unusual in littoral environments where energetic coastal processes normally result in well-sorted beach deposits (McCave and Syvitski, 1991; USACOE, 1984). It is unlikely that these apparent differences in mud fractions reflect a genuine change in environmental conditions during 1997. If that were the case, one would expect significant differences in other size fractions as well. Instead, average size fractions were similar within the sand and gravel range (Figure 9) where method comparability is greatest (Appendix A.4).

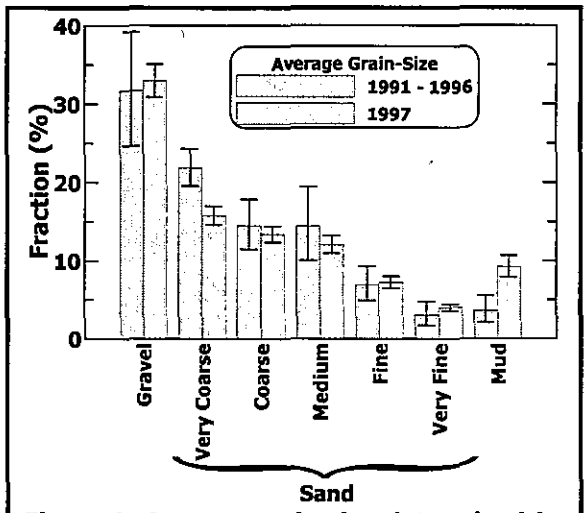


Figure 9. Average grain-size determined by volumetric analysis (before 1997) and by weight (in 1997) along with 90% confidence intervals

Results

As described in the introduction, ongoing recovery manifests itself in a time sequence of mean abundance at impacted sites that significantly departs from that of control sites. Its statistical significance is measured by testing the null hypothesis that mean time sequences at control and

impacted sites are parallel. If the significance level is small ($p<0.10$), then there is less than a 10% chance that differences in temporal sequences could have arisen if the time profiles were actually parallel. Thus, lower p-values indicate greater confidence that recovery is occurring within the time window. In contrast, higher p-values may indicate that either recovery has occurred or that the statistical test could not detect more gradual ongoing recovery because of low statistical power. Thus, higher p-values do not necessarily mean that the time profiles are parallel; they simply indicate that any difference in trends was too small to easily detect with the given sample size. Because sample sizes are small (four degrees-of-freedom) for 6-year test windows, only very large departures from parallelism will be fully resolved by the tests. Nevertheless, the relative amplitude and timing of recovery processes can be determined by comparing p-values in adjacent time windows.

Total Infaunal Abundance

Total infaunal abundance recovered rapidly between 1991 and 1992. This recovery event is evident as a marked increase in the mean abundance at Category-2 and 3 sites (Figure 1 of the Executive Summary). A similar increase did not occur at control sites indicating that the change at impacted sites was not in response to regional changes in ambient environmental conditions. In fact, the rapid recovery in 1991-1992 at Category-3 sites was large enough to be detected because the p-value of parallelism test was 0.05, which is below the prescribed significance level of $\alpha=0.10$. Figure 10 displays the results of the three separate parallelism tests conducted on data within the three available 6-year windows. Parallelism test results will be presented in a similar manner throughout this report. The upper portion of the figures display the time profile of logarithmic abundance averaged over control sites (Category 1) along with the average time profile over a group of impacted sites, either Category 2 or Category 3. The bottom portion of the figure shows the results of the parallelism tests conducted on data within a time window whose span is indicated by the horizontal bar. The p-value of the test is indicated by the height of the bar relative to the ordinal axis shown in the lower right of the figure. The lower the bar, the more likely that significant recovery is ongoing within the time window.

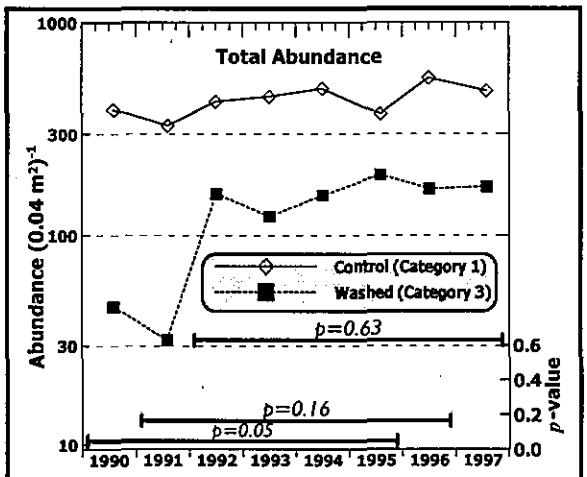


Figure 10. Parallelism tests on Category-3 mean infaunal abundance

Thus, the first test window in Figure 10 clearly captures the abrupt repopulation event that only occurs at the impacted (Category-3) sites. Although not statistically significant, the adjacent window also exhibits a low p-value (0.16) because it encompasses one year (1991) of impacted abundance. In contrast, full stabilization of infaunal abundance at Category-3 sites between 1992 and 1997 is particularly apparent within the last 6-year window. For the abundance time profiles encompassed by that window, there is little reason to reject the null hypothesis of strong parallelism (p=0.63). Had the recovery process been less abrupt and spanned a period of several years, then the sequence of increasing p-values would have also been more gradual.

A similar sequence of test results was obtained for the comparison between Category-2 sites and controls. However, the Category-2 abundance did not increase as dramatically in 1992 (Figure 1). Consequently, the null hypothesis of parallelism could not be rejected ($p \geq 0.19$) even in the initial 6-year window. Nevertheless, ongoing recovery of some individual Category-2 species was detected with statistically significant departures from parallelism at the 10% (α=0.10) significance level. Species that measurably participated in the recovery at Category-2 sites are described in the subsequent section on individual taxa.

Credence in the overall pattern of impact and recovery is reinforced by the consistent time profiles at individual sites within each category. The time profiles of the Category-3 sites exhibit the clearest recovery signature (Figure 8). In 1990 and 1991, the abundance at all three sites included in the Category-3 mean were markedly lower than in 1992 and following years. Individual Category-2 sites all showed a similar consistent pattern of early increases in abundance although infaunal repopulation at Mussel Beach was more gradual and extended through 1994 (Figure 7).

Major Taxonomic Groups

The recovery of total infaunal abundance between 1991 and 1992 was a major event within the infaunal community as a whole. The abrupt increase in total infaunal abundance could have resulted from large changes in the population of a single taxon. Instead, it pervaded the infaunal community and encompassed diverse taxonomic groups including annelids, mollusks, and other infaunal phyla. In the case of mollusks, the amplitude of the repopulation was large enough to be detected by parallelism tests. The recovery signature of phyla other than annelids, mollusks, and crustaceans was weaker and began earlier in 1990. Also, crustacean recovery was not clearly defined due to variability in control populations. Other than total crustacean abundance, the taxonomic breadth of the repopulation event emphasizes its importance.

Time profiles of annelid and mollusk abundance (Figure 11 and Figure 12) exhibit the distinctive signature of recovery at both Category-2 and 3 sites. As with total infaunal abundance, control populations were comparatively uniform throughout the monitoring program. Also, the amplitude of the abrupt 1991-1992 repopulation was larger for washed sites (Category 3) than for oiled sites where intensive cleanup methods were not applied. As before, this suggests that impacts to both annelids and mollusks from hydrocarbon exposure and habitat disturbance were more severe than from the chemical contamination alone.

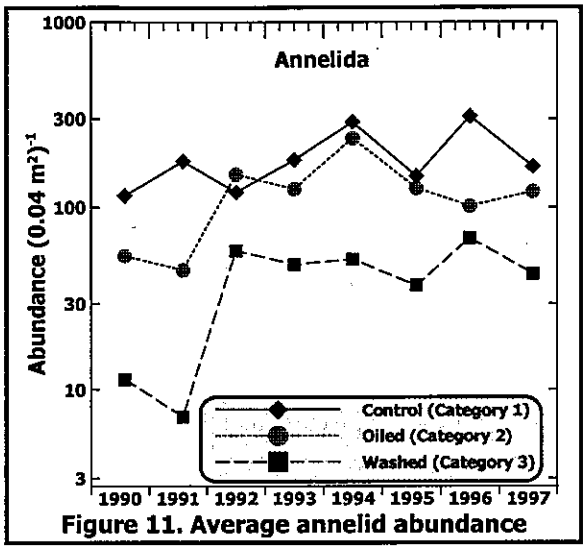


Figure 11. Average annelid abundance

Time profiles of the combined abundance of infauna other than annelids, mollusks, and crustaceans, also exhibit a recovery signature (Figure 13). However, their overall abundance was lower (note the scale shift in the vertical axis) and average control populations were comparable to recovered populations at Category-3 sites. Also, repopulation at washed sites (Category 3) appears to have begun a year earlier in 1990. This suggests that different components of the intertidal community may have recovered at slightly different times.

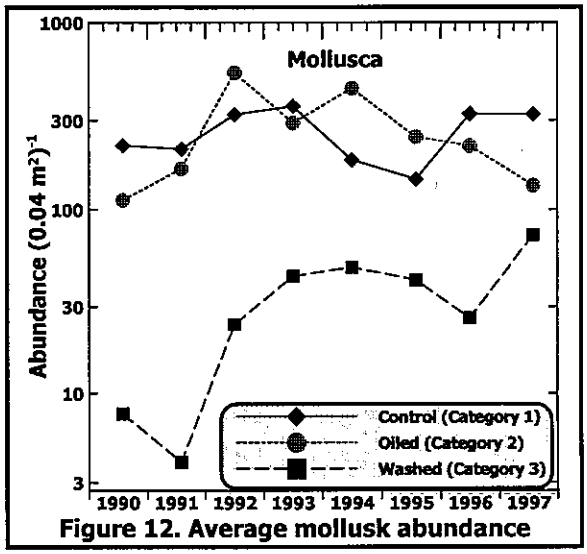


Figure 12. Average mollusk abundance

Although the crustacean time profiles at impacted sites exhibit a sharp population increase in 1991 (Figure 14), recovery of this taxonomic group is not as clearly defined as other groups. The high interannual variability in crustacean abundance masks the recovery patterns when impact and control time profiles are compared. Specifically, control sites exhibited low mean abundance in 1990 and 1991 so their time profiles initially parallel those of impact sites. However, some of this

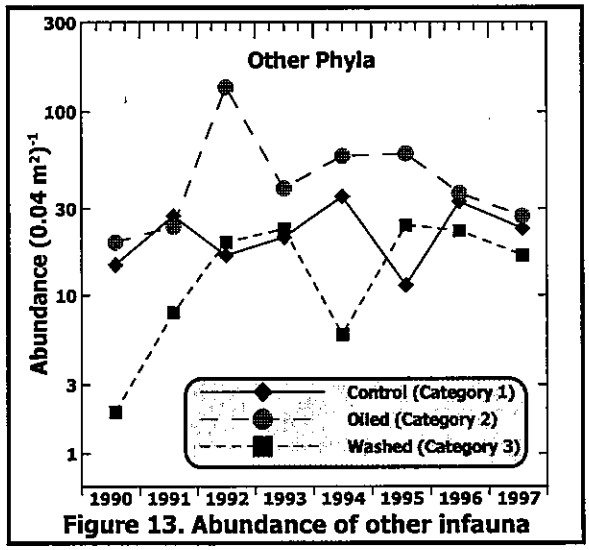


Figure 13. Abundance of other infauna

interannual variability may be an artifact of temporal inconsistency in taxonomic identification and enumeration. Reexamination of the entire set of crustacean specimens in a manner similar to that conducted on the mollusk data could resolve much of the interannual variability and reveal a clearer pattern of impact and recovery. Irrespective of this, some individual species of crustaceans exhibited the distinctive pattern of impact and recovery as discussed in the following section.

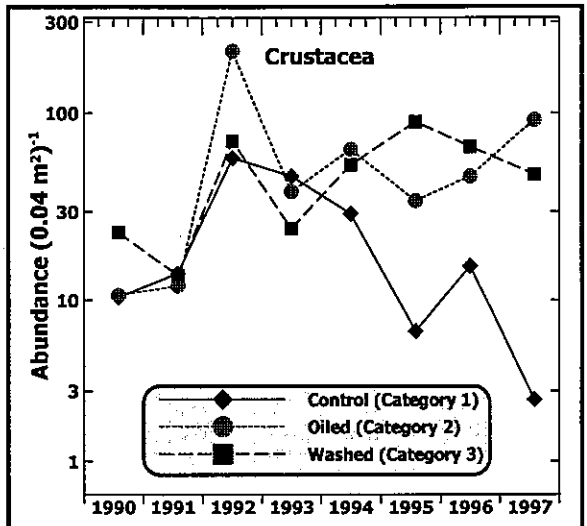


Figure 14. Average crustacean abundance

Because of the large increase in the mollusk population between 1991 and 1992, ongoing recovery at Category-3 sites was detected by statistically significant departures from parallelism in the first of the three test windows (Figure 15). As with total infaunal abundance, the *p*-value within the second window (1991-1996) was low, but not quite statistically significant (*viz.*, ≤ 0.10). Time profiles within the last test window (1992-1997) had a high degree of parallelism although the time profile at washed sites exhibited a curious arch structure with a secondary population increase between 1996 and 1997.

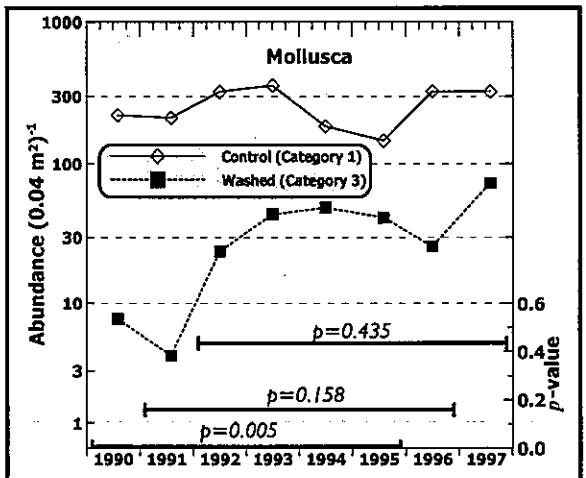


Figure 15. Parallelism tests on mean molluscan abundance

This Category-3 post-recolonization time profile is similar to that of rockweed (*Fucus gardneri*) shown in Chapter 3 (see Figure 28b) and discussed by Houghton *et al.* (1997a). As with rockweed, the mollusk profile may reflect the senescence (aging) of a single cohort (year-class) of organisms that were recruited during the abrupt repopulation event between 1991 and 1992. Additional mollusk recruitment results in continued gradual population increases between 1992 and 1994. Between 1994 and 1996, this recruitment is offset by senescence of the large cohort initially recruited between 1991 and 1992. The population increase in 1997 may represent a secondary recruitment. If this age-class model applies to both mollusks and rockweed, then the nominal molluscan longevity is about four years and is similar to that of *Fucus gardneri*. In fact, as discussed in Chapter 3, a wide range of intertidal taxa exhibit a similar arch in their Category-3

time profiles and it is unlikely that all of these organisms have similar life-history characteristics. An alternative explanation is that the observed mollusk time profiles result from a trophic interaction between herbivorous molluscan grazers and rockweed cover. Still another is that the arch is part of damped oscillations that arise naturally as part of a feedback mechanism in simple population models.

Regardless of our ability to characterize these smaller amplitude fluctuations in terms of population dynamics, they cannot be validated statistically with parallelism tests. The relative increase in mollusk populations after 1992 was only about half of the initial increase and was comparable to natural fluctuations in control populations. Also, the time-span of the arch is too short for parallelism tests to separate gradually increasing or decreasing population trends using the six-year time windows (Figure 15). Instead, strict interpretation of parallelism tests indicates that a broad range of impacted taxonomic groups had largely recovered by 1992 and that populations remained comparatively stable in subsequent years. It is only through continued monitoring that the secondary fluctuations in populations can be better described.

Individual Taxa

As with major taxonomic groups, the time profiles of individual taxa also disclose the distinctive pattern of impact and recovery; namely, low initial abundance at impacted sites that abruptly increased after 1991 for some taxa. This increase was not evident at control sites and was followed by a period of relative stability. While the time histories of dominant organisms may be expected to closely track total abundance, less prevalent species also played a role in overall recovery of the infaunal community. Instead of examining the time profile for each of the 121 infaunal taxa, a principal component analysis (see Appendix A.1) was applied to identify those organisms that had the clearest signature of recovery. Because there was no reason to assume that the same set of taxa would influence recovery at both categories of impacted sites, separate principal component analyses were performed on the data from oiled (Category-2) and washed (Category-3) sites.

The biplot shown in Figure 16 provides a visual interpretation of the principal component analysis that was performed on the Category-3 abundance data. It is called a biplot because it simultaneously displays the similarity among infaunal samples (by the proximity of the points) and the species that differentiate the samples (shown as arrows or vectors). Thus, the biplot arrows help identify those species exhibiting the strongest pattern of recovery at Category-3 sites. Analogously, differences in infaunal communities among Category-3 samples is reflected by the distance between the symbols (■) on the biplot.

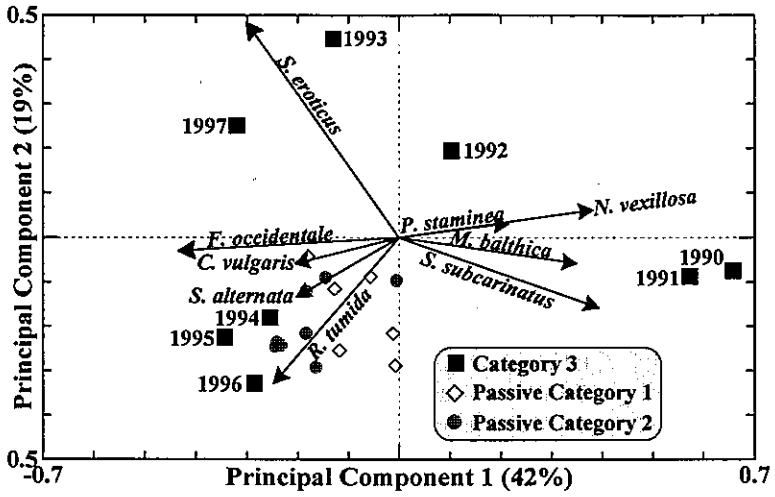


Figure 16. Principal infaunal components at Category-3 sites

The distribution of the Category-3 samples along the first principal component (x-axis) of the biplot successfully captures the 1991-1992 repopulation event at Category-3 sites. This axis accounted for a large portion (42%) of dissimilarity among Category-3 samples and clearly showed that the community structure within 1990 and 1991 samples was distinctly isolated from that of more recent samples. Specifically, these early samples had large positive principal components that transitioned through the small (0.1) positive component of the 1992 sample, to negative values in recent years. Thus, the infaunal community within Category-3 samples collected in 1990 and 1991 were comparable to one another but bore little resemblance to the community within the other more-recent samples.

Before examining which taxa were most responsible for these temporal differences, it is useful to examine how samples from Category-1 and 2 sites compare with the Category-3 samples. This can be accomplished after Category-3 ordination by plotting the location of these passive samples (shown by ● and ◇ symbols) on the biplot by using the same weighting of species that determined the location of the Category-3 points. Because they do not materially participate in the ordination (*viz.*, in the determination of species weights), they are designated '*passive*' samples. It is comforting to note that their (less-impacted) infaunal communities are most similar to those of the recent (more-recovered) Category-3 samples as indicated by their entirely negative x-axis components.

The taxa that materially participated in the post-1991 recovery event were identified using the biplot arrows in Figure 16. Once identified, parallelism tests were conducted on time profiles for each taxon to assess the statistical significance of its repopulation relative to control samples. Before discussing the results of these tests, the selection of candidate taxa based on the interpre-

tation of biplot arrows is described. One of the precepts of biplots (Appendix A.1) is that the arrows point in the direction of samples with comparatively increased abundance of a given species. Thus, arrows that point toward the left (away from the 1990 and 1991 samples) represent the species of primary interest. The ordination indicates that they were comparatively rare in samples collected in 1990 and 1991 but their abundance increased substantially after 1991. This recovery pattern is consistent with the time profiles of total infauna and major taxonomic groups that were discussed previously.

Although all of the taxa identified by negative vectors in Figure 16 have populations that increased after 1991, they do not all necessarily represent a significant departure from parallelism with control populations. Recall that the ordination was conducted on Category-3 sites alone and that the passive location of Category-1 sites was shown for comparison. Thus, the orientation of the vectors does not include temporal information about changes in control population. In fact, there was a temporal increase in control populations of *Rochefortia tumida* (a small <4 mm infaunal bivalve) that tracked the Category-3 increase between 1991 and 1995. Thus, the resulting trends for this taxon did not significantly depart from parallel (p>0.10 in Figure 17).

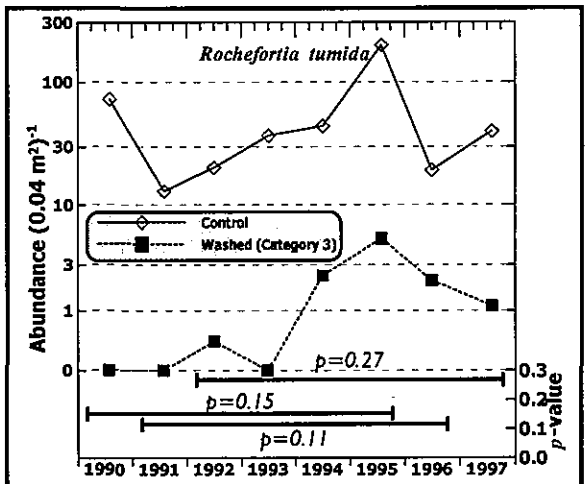


Figure 17. Parallelism tests on Category-3 abundance of the small but prevalent bivalve *Rochefortia tumida*

This was not true of three other species that included a small (≤ 1 cm) shrimp-like crustacean *Cumella vulgaris*, a small shallow-water detritus-feeding gastropod *Fartulum occidentale*, and the highly motile carnivorous polychaete *Syllis alternata*. All participated in the recovery at Category-3 sites even though they were not the most abundant infauna. Their biplot arrows point away from the 1990 and 1991 samples indicating that they were comparatively rare in these early samples. All three species exhibited statistically significant increases in abundance at Category-3 sites relative to control sites and two are shown in Figure 18 and Figure 19.

Their time profiles indicate that these taxa recovered at a slower rate and stabilized later than that reflected in time profiles of total abundance. Populations of the detritus-feeding gastropod *F. occidentale* and the carnivorous polychaete *S. alternata* did not stabilize until around 1994 and consequently, the first two test windows exhibited significant departures from parallelism. This

contrasts with time profiles of total infauna and mollusks where significant departures from parallelism were only observed in the first test windows.

Although they are not shown here, the time profiles of the shrimp-like crustacean (*C. vulgaris*) departed significantly from parallelism in all three test windows. However, this was an artifact of steadily declining control populations rather than continuously increasing Category-3 populations. Two other species, the deposit-feeding burrowing polychaete *Ophelia limacina* and the leafy paddle worm *Eteone longa*, also exhibited statistically significant departures from parallelism at Category-3 sites, although they did not strongly influence the multivariate analysis.

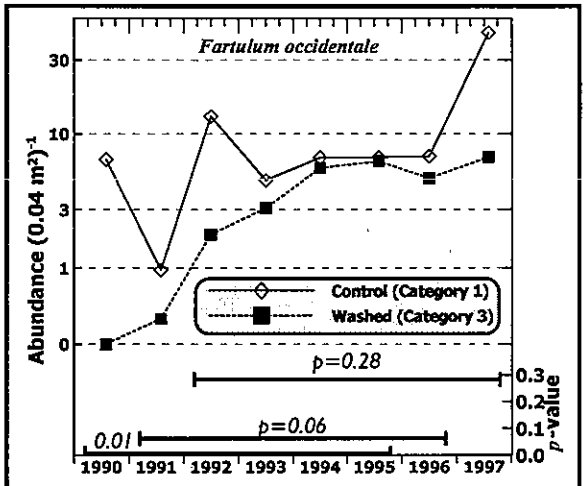


Figure 18. Parallelism tests on Category-3 abundance of the small detritus-feeding gastropod *Fartulum occidentale*

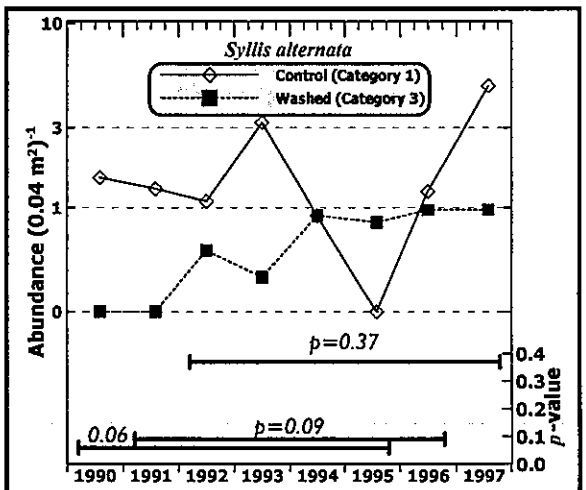


Figure 19. Parallelism tests on Category-3 abundance of the motile carnivorous polychaete *Syllis alternata*

Principal component analysis of Category-2 (oiled but not washed) infaunal data reveals similar temporal trends indicative of recovery although a different suite of taxa exhibited markedly increased abundance after 1991 (Figure 20). Again, the unique character of the infaunal community in 1990 and 1991 separates the Category-2 samples (shown by ● symbols) along the first axis. This axis accounted for a large portion (38%) of dissimilarity among Category-2 samples. In contrast to the Category-3 biplot (Figure 16), passive control samples (shown by ◇ symbols in Figure 20) are located between the x-axis extremes of the Category-2 samples. This suggests that the Category-2 infaunal communities, which were exposed to oil but were not subject to intensive cleanup methods, were less impacted than Category-3 communities whose structure in 1990 and 1991 was markedly different from that of control sites. This would also account for the greater increase in total infaunal abundance that was observed at Category-3 sites after 1991, as compared to the repopulation at Category-2 sites (Figure 1).

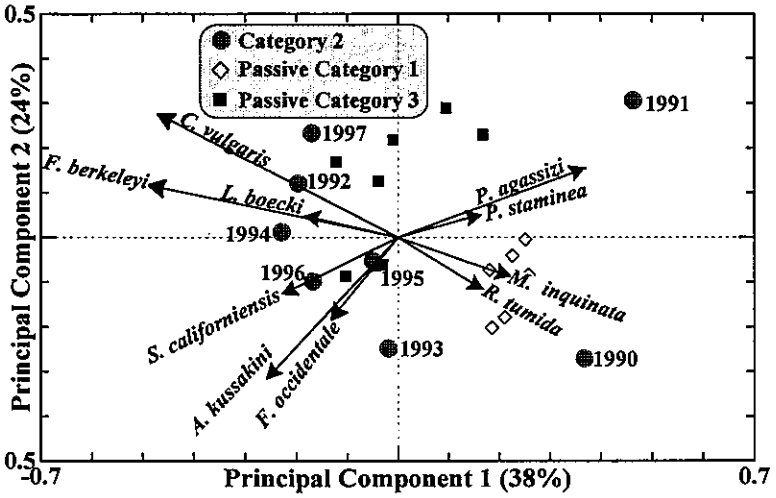


Figure 20. Principal infaunal components at Category-2 sites

Only two of the species shown by biplot arrows in Figure 20 exhibit a significant lack of parallelism in their time profiles (*Fabriciola berkeleyi* and *Laphania boecki*). As with Category-3 taxa, biplot arrows for these two tube-building polychaetes extend along the negative x-axis and away from the 1990 and 1991 samples. This reflects a marked increase in their abundance after 1991. The time profile for the filter-feeding sabellid worm *Fabriciola berkeleyi* displays a distinct recovery event at Category-2 sites between 1991 and 1992 (Figure 21). As a result, only the first test window displays a statistically significant (p=0.07) departure from parallelism. This is consistent with the onset of recovery observed in total infaunal abundance and annelids (Figure 11) at both Category-2 and 3 sites. In contrast, the terebellid polychaete *Laphania boecki* began recovering later, in 1993, and all three test windows exhibited significant departures from parallelism ($p \leq 0.09$ in Figure 22). In addition to these taxa, the small shallow-water detritus-feeding gastropod *F. occidentale* exhibited a statistically significant (p=0.006) departure from parallelism at Category-2 sites, in addition to its role in recovery at Category-3 sites.

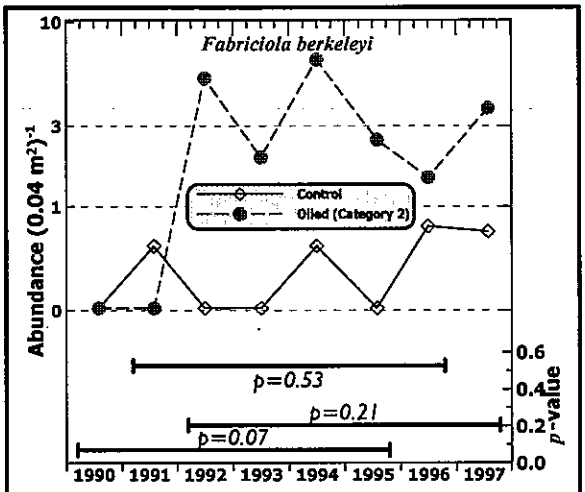


Figure 21. Parallelism tests on Category-2 abundance of the tubicolous filter-feeding polychaete worm *Fabriciola berkeleyi*

Sediment-Infauna Interaction

The foregoing analyses demonstrate that infaunal abundance at impacted sites increased abruptly over a brief one-year period after 1991. Most of the infaunal community stabilized by 1992 although the abundance of some impacted taxa did not stabilize until 1994. Infauna at the control sites did not exhibit a concomitant pattern of increasing abundance. In comparison, the infaunal abundance at control sites was relatively stable throughout the monitoring program. However, the Category-3 infauna stabilized at an abundance level that was well below that of the control and Category-2 sites. Moreover, the Category-3 sites supported a post-recovery (stabilized) infaunal community that was quite different from most other monitoring sites.

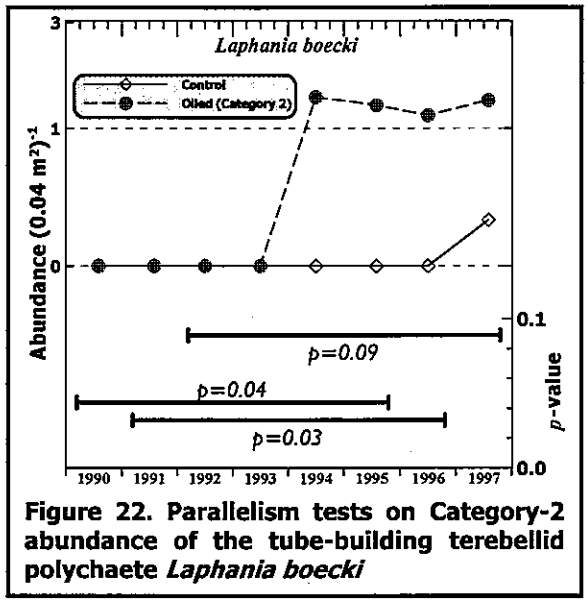


Figure 22. Parallelism tests on Category-2 abundance of the tube-building terebellid polychaete *Laphania boecki*

There are two possible reasons for the difference in the stabilized community at the Category-3 sites. First, the distinct infaunal community could arise naturally from inherent differences in the physical environment at the three sites that happened to be included in the Category-3 average. If these environmental differences were present before the spill, then the Category-3 community has fully recovered to pre-spill conditions. Alternatively, the infaunal community that was subjected to hot-water washing may not have reached full equilibrium with the environmental conditions at Category-3 sites and is continuing to slowly recover. This second hypothesis can be resolved with continued long-term monitoring which would allow parallelism tests to better resolve subtle temporal trends. However, a canonical analysis of the current database demonstrates that infauna at Category-3 sites have achieved equilibrium with present environmental conditions; and that those conditions, namely grain size, consistently differ from those conditions at Category-2 and 3 sites.

Canonical analysis (see Appendix A.1) extends standard multivariate techniques by relating infaunal variability to environmental conditions. Moreover, the significance of relationship between infauna and environmental parameters can be tested directly, without the need for restrictive assumptions (*e.g.*, independence) concerning the applicability of a specific probability distribution (*e.g.*, normality). Instead, samples are randomly reassigned (permuted) to environ-

mental parameters to see how often a level of correspondence similar to the original one can be achieved by chance alone. These Monte Carlo permutation tests thus provide a direct measure of the significance probability or *p*-value.

Monte Carlo tests were applied in a canonical correspondence analysis (CCA) between physicochemical parameters (grain size, TOC, TKN) and infaunal data collected at all the lower intertidal monitoring sites between 1993 and 1997. During this period of stable infaunal communities, TOC and TKN concentrations within sediments were not significantly related to the infaunal abundance. This is reflected by their high *p*-values (p=0.64 and p=0.38, respectively) which indicate that a similar degree of correlation could occur by chance alone at least one out of three times.

Conversely, the Monte Carlo tests revealed a highly significant ($p \leq 0.001$) relationship between infauna and the 63-μ sediment fraction (very fine sands). In fact, variability in all medium-to-fine sand fractions (63 μ to 0.5 mm) was closely linked to the distribution of infauna among samples ($p<0.04$). However, this is expected given the high multicollinearity among these fractions. In other words, samples that were enriched in fine sands tended to have an increased amount of medium sand as well. This was not the case for the very coarse sand fraction (1 mm to 2 mm) which was unrelated to variation in the other sand fractions. Nevertheless, it also explained a significant amount of additional infaunal variation among samples (p=0.08 after accounting for influence of medium-to-fine sands). Finally, after accounting for all the sand fractions, the Monte Carlo tests found that the larger gravel (>2 mm) fractions had little influence on infauna (p=0.20). Thus, these permutation tests demonstrated that only two of the measured physicochemical parameters strongly influence infaunal distribution, namely coarse and medium-to-fine sand fractions.

Canonical correspondence analysis (CCA) relates three types of variables and output is in the form of a triplot rather than the biplot shown previously in Figure 20. In a triplot (Figure 23), samples and species are both represented by points whereas the environmental variables are indicated by vectors. The qualitative interpretation is generally the same; namely, a vector points toward samples with increased levels of a particular environmental parameter. Thus in Figure 23, samples located further along the negative y-axis are relatively enriched in very fine sands. Analogously, samples with positive y-values lack fine sands. This triplot also simultaneously shows the species (by ★ symbols) that constitute the primary differences in communities among samples. Their proximity to a particular set of sample points indicates that they are relatively abundant in those samples.

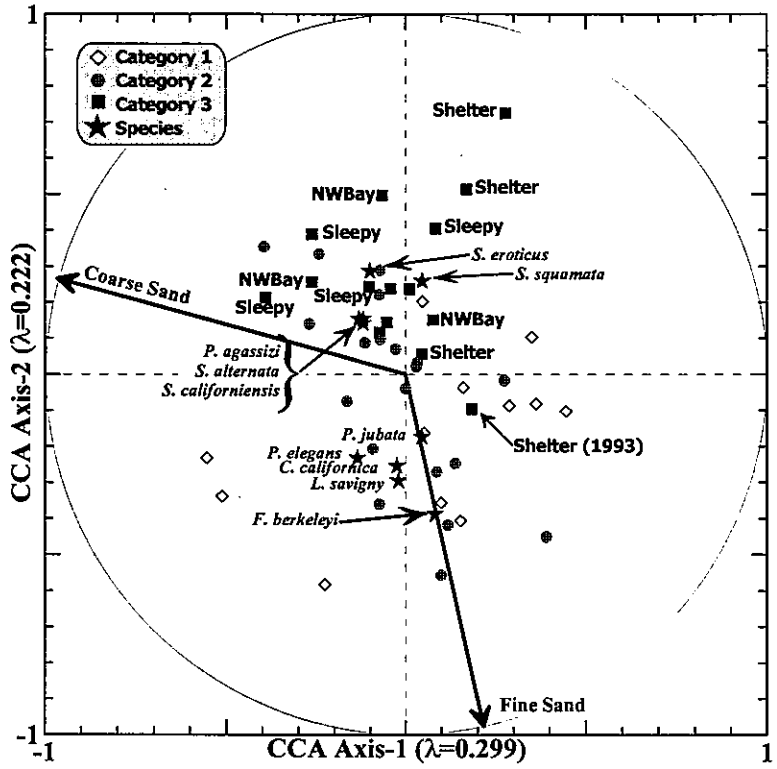


Figure 23. Species-conditional triplot displaying the canonical correspondence between grain size, infaunal species, and sampling site

The deficiency of fine sands at Category-3 sites is a striking feature of Figure 23. Category-3 samples are represented by ■ symbols and except for the sample collected at Shelter Bay in 1993, all Category-3 samples had below-average amounts of fine sediments. This is evident from their positive y-axis locations, which are positioned opposite of the gradient of increasing fines indicated by the vector. In contrast, samples from other categories encompass a broad range of grain-sizes including some samples that were comparable to the Category-3 samples both with respect to their lack of fines and their associated infauna.

Thus, this correspondence analysis shows that the observed differences in stabilized infaunal populations at Category-3 sites is expected given the absence of fine sediments at those sites. Upon recolonizing of Category-3 sites between 1991 and 1992, the infaunal community achieved equilibrium with the ambient grain-size distribution. Consequently, the second hypothesis that infaunal differences at Category-3 sites reflect incomplete equilibration with ambient environmental conditions, can be rejected. On the contrary, this report shows that any major future changes in Category-3 infauna will not be the result of recovery driven by a continuing imbalance between the community and its environment. Furthermore, infauna at Category-3 sites will

not asymptotically approach mean control communities unless the physical environment changes.

Because of this, recovery assessments based solely on convergence in absolute population levels at control and impact sites are flawed if environmental conditions at Category-3 sites were different prior to the spill. If instead, hot-water washing of Category-3 sites during cleanup removed a significant amount of fine sediments, then infaunal communities have not recovered to pre-spill conditions despite their apparent stability. Thus, analysis of the current database cannot unequivocally resolve whether the stabilized infauna have fully recovered to pre-spill conditions. Only continued monitoring of the form described in the following section will indicate whether subtle population shifts are continuing to occur in response to slow redeposition of fine sediments at Category-3 sites.

Before discussing future monitoring, it useful to identify the species that were influential in differentiating samples with different grain-size distributions. The CCA triplot (Figure 23) displays these influential species along with the sample distribution and environmental trends that can be used to interpret observed shifts in community structure. The tubicolous filter-feeding sabellid polychaete *Fabriciola berkeleyi* was only present in samples containing a substantial amount of fine sand. This explains why it did not play a role in recovery at Category-3 sites, although it contributed strongly to the recovery process at Category-2 sites (Figure 21). In contrast, two annelids, the carnivorous archiannelid *Saccocirrus eroticus* and the widespread tubicolous spionid *Scolelepis squamata*, were closely associated with the coarser sediments. Although neither played a strong role in the recovery process at Category-3 sites, they characterize the distinct infaunal assemblage within the coarser sediments of Prince William Sound. Similarly, sediments with slightly below-average amounts of fine sand contained an increased abundance of two syllid polychaetes, the motile carnivore *Syllis alternata* and selective deposit feeder *Sphaerosyllis californiensis*, as well as the peanut worm *Phascolosoma agassizi*. The carnivorous syllid *S. alternata* was one of the organisms whose abundance increased significantly at Category-3 sites relative to controls.

Recommendations

Continued intertidal monitoring within Prince William Sound will lend valuable additional insight into the recovery infaunal organisms and the role that invasive cleanup techniques played in that recovery. However, given the results described in this chapter, a slight redirection in the monitoring effort would better serve to address the remaining issues. We now know that a broad assemblage of infauna experienced an abrupt increase in abundance at both categories of im-

pacted sites. The community subsequently stabilized in equilibrium with the controlling environmental factor, namely fine-sediment content. The monitoring effort should now be directed toward increasing the power to detect much subtler temporal trends in both infauna and grain size. The following recommendations are made with that in mind.

- Continue infaunal monitoring at the nine core sites listed in Table 2. For example, if the field sampling effort is limited during future surveys, collection of samples from these nine sites should take precedence over sampling at other lower-intertidal sites. These sites have been consistently sampled on an annual basis throughout the program, including those years prior to the distinct infaunal recovery event of 1991-1992. If infauna are continuing to gradually recover at Category-3 sites, future samples will provide valuable degrees-of-freedom for increasing the power to test parallelism. Because of their temporal continuity, the nine sites act as sentinels for any ongoing long-term recovery. If resources are available, monitoring at additional control sites would better define natural population fluctuations that result from regional influences within Prince William Sound. This, along with an improved estimate of inherent spatial variability among sites, would further enhance the ability to detect shifts in impacted infaunal communities.
- Expand sampling of physicochemical parameters, specifically grain size and interstitial salinity. The strong relationship between grain-size and infauna has been described in the foregoing analyses. In addition, anecdotal observations suggest that freshwater streams occasionally impinge on infaunal sampling sites. Like grain-size, large changes in salinity can profoundly impact marine organisms. Precise monitoring of these influential environmental properties can aid in the interpretation of infaunal variability, particularly when the goal is to distinguish anthropogenic (human-induced) impacts from natural trends.
- Collect grain-size samples and salinity measurements at all replicate infaunal sampling locations within a particular site. This would provide insight into processes that drive small-scale infaunal variability within sites. Currently, only one sediment sample is collected from each site for analysis of physicochemical properties. If the fine sediment distribution within sites is highly heterogeneous, then a single grain-size sample is probably not representative of the site. As a result, the importance of the observed grain-size differences between sites would be diminished. This determination cannot be made from the replicate grain-size samples that were collected during the 1995 survey. Volumetric analysis of those replicate samples was incapable of accurately determining the finest sediment fractions. Application of a modified Plumb (1981) technique to future replicate samples would precisely measure the fine-sediment fractions. Contemporaneous salinity measurements can be used to further refine the grain-size analysis by accounting for the weight of dissolved solids (salts) during pipette analysis (Appendix A.4).
- Conduct manipulative experiments to address any remaining uncertainty concerning the degree of ongoing recovery at Category-3 sites. Carefully designed experiments involving hot-water washing of limited areas within control sites could directly assess the infaunal and physicochemical impacts from the invasive shoreline treatment techniques that were applied in 1989. Alternatively, fine-grained sediment could be artificially introduced into portions of Category-3 sites to monitor infaunal repopulation and sediment dispersal. As a companion to

this, various coastal process studies could be conducted to determine whether the lower intertidal zone of Category-3 sites are naturally dispersive with respect to fine sediments.

CHAPTER 3. INTERTIDAL EPIBIOTA

Introduction

Intertidal epibiota are organisms that live on or grow directly attached to hard substrata that lie between the highest and lowest reaches of the tides. This section examines the recovery of epibiotic assemblages within Prince William Sound that were impacted by the *Exxon Valdez* oil spill in 1989. Dominant epibiota within the Sound's intertidal zone consist of rockweeds, barnacles, mussels, and littorine and limpet snails. While algae are the dominant plant forms, marine lichens can also occur in the uppermost elevations. Most intertidal species are restricted to certain elevations or tidal zones. Upper limits on the distribution are generally determined by a species' ability to withstand desiccation, while lower limits are largely defined by grazing, predation, competition for space, and habitat preferences.

Because they are only occasionally wetted, the rock surfaces within the upper-intertidal zone support a less-diverse community and are usually barren or only sparsely covered with algae. The occasional intertidal pool within the upper zone will contain species more commonly found within lower-elevation habitats. Generally, however, upper-level epibiota consist mostly of patches of invertebrates such as the grazing periwinkle snail *Littorina*, which typically occurs in rock crevices, and *Verrucaria*, a marine lichen.

In the middle intertidal zone, the brown rockweed *Fucus gardneri* is abundant. It often forms a canopy beneath which small filamentous and fleshy red algae occur. Dense aggregations of the filter-feeding barnacles, *Balanus* and *Semibalanus*, and the mussel *Mytilus* cf. *trossulus*, can also reside beneath the *Fucus gardneri* canopy. Grazing limpets (*Lottia*), and predators such as drills (*Nucella*), scavenger hermit crabs (*Pagurus*), and seastars (*Leptasterias*), also inhabit the middle intertidal zone.

In the lower intertidal zone, red algae are generally more abundant than at middle elevations. The increased invertebrate diversity within this zone also results from the presence of tube-building species of worms, encrusting sponges, and tunicates. Lower intertidal areas with large, stable rock habitats support more diverse epibiotic assemblages than substrata consisting of shifting cobble, gravel, and sand.

Methods

Field Methods

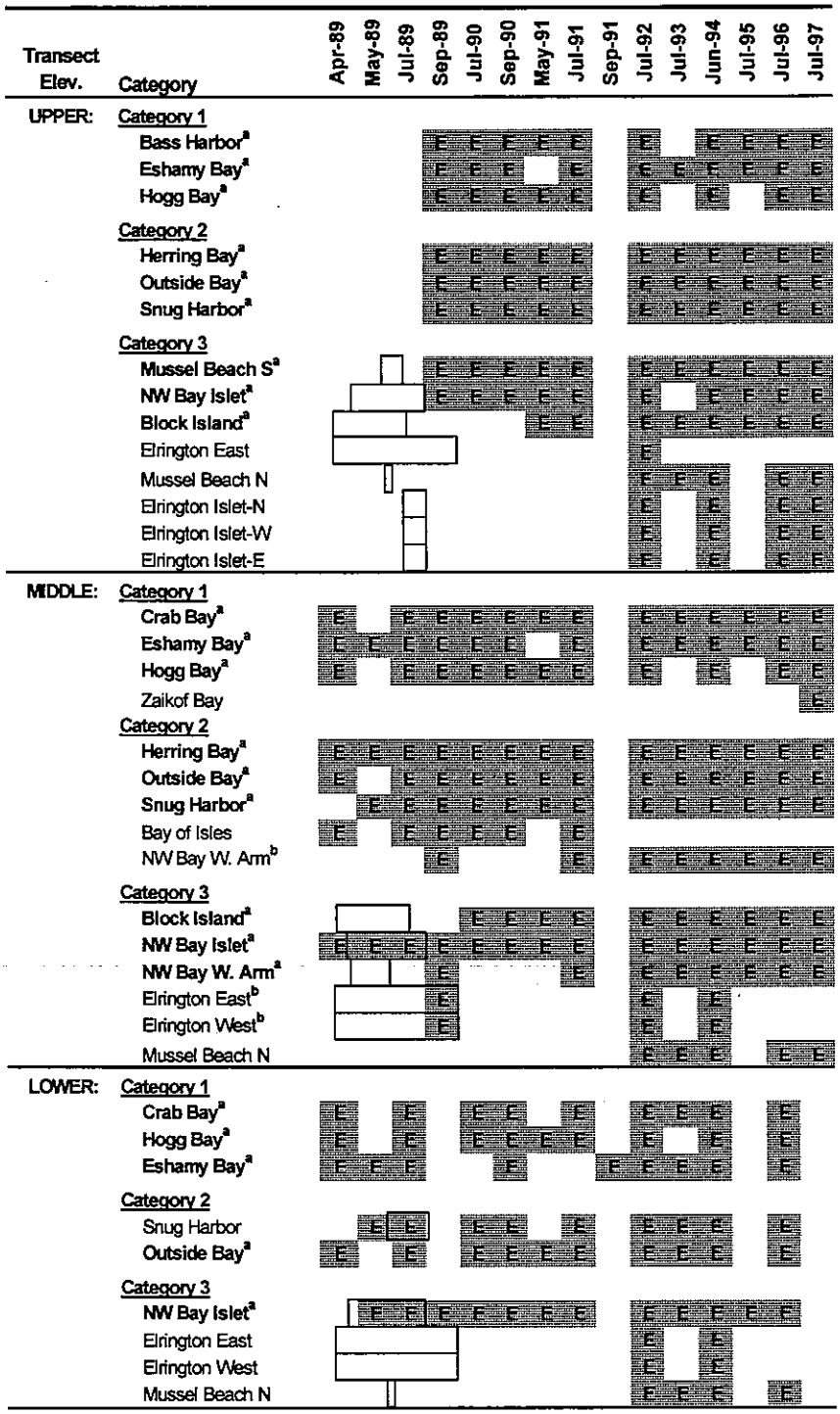
Epibiota were sampled at sites that were classified into the three categories described in Chapter 1. However, in contrast to infaunal sampling where only one tidal elevation was consistently sampled, epibiota samples were enumerated at multiple tidal elevations (Table 3). Consequently, categorization was based on the history of oil exposure and shoreline cleanup method applied at the tidal elevation where sampling occurred. Epibiotic sampling was conducted within five (at upper elevations) or ten (at middle elevations) permanent 0.25 m^2 quadrats established along a transect parallel to the shoreline. Transect elevations were measured and an index of exposure to environmental conditions was estimated at each site. Transect and quadrat locations were initially established in 1989 as described in Houghton *et al.* (1993b). Since 1989, sampling sites have occasionally been added and in 1997, Zaikof Bay was added as a new control site. The chronology of epibiota sampling is shown in Table 3. Historically, epibiota has been sampled along three transects at each site. However, sampling at the lower tidal elevations has been discontinuous throughout the monitoring program due to limited access except during extreme low tides. In view of this, no epibiotic sampling was attempted at the lower tidal elevation in 1997.

Epibiotic enumeration within each quadrat consisted of counts of individual organisms and species coverage estimates (Appendix C). All species were identified to the lowest taxonomic level possible. Percent cover was estimated for overstory, understory, and crustose algae, as well as for substrate cover, attached invertebrates such as mussels and barnacles, and encrusting invertebrates such as sponges and tunicates. Species whose coverage was less than one percent were recorded as '*trace*' and were assigned a default value of 0.5% in numerical analyses. The total of plant, invertebrate, and bare substrate cover often exceeded 100 percent within a quadrat due to multilayering of species. Motile invertebrates were counted for number of individuals. When apparent, dead organisms were noted and organisms that were difficult to identify were grouped into broader taxonomic categories. Organisms that could not be identified in the field were collected from outside the quadrats and identified in the laboratory.

Analysis Methods

An assessment of epibiotic recovery was conducted in the same manner as for infauna, namely a regression of Equation 2 within moving 6-year time windows. The same reasoning applied for testing parallelism rather than convergence between impact and control sites. Namely, impacted sites (*i.e.*, Category 2 and 3) probably supported epibiotic populations that were different from

Table 3. Epibiotic rocky transects sampled from 1989 through 1997



☐ Approximate time windows in which treatments occurred in 1989.

[a] Core group of stations used in calculation of category means.

[b] Category uncertain due to unknown cleanup history.

control populations prior to the spill. These biological differences resulted from systematic environmental differences between control and impact sites. Consequently, the category mean of impacted community indices may never be expected to '*recover*' to the same absolute amplitude of control sites.

As with the assessment of infaunal recovery, this chapter focuses on data from a subset of core sites that were consistently sampled over the course of the monitoring program (Table 3). Unlike the infaunal database, epibiotic taxonomic identifications were not re-examined on a global basis. Also, in contrast to infaunal data, the epibiotic database included multiple tidal elevations, several distinct biotic assemblages, and an additional year (1989) of data. Consequently, separate analyses were performed on three assemblages and two tidal elevations. Only the upper and middle intertidal elevations were examined for epibiotic recovery. Lower intertidal elevations were excluded from the analyses because they were not regularly sampled at multiple sites within each category and because no lower intertidal sites were sampled in 1997.

Recovery in the three epibiotic assemblages, that is, algal cover, invertebrate density, and invertebrate cover, was also separately evaluated. Fundamental biological differences among these assemblages suggest that they may be impacted and recover in distinct ways. To further narrow the focus, analysis of algal cover excluded crustose taxa. By necessity, invertebrate abundance and percent cover were analyzed separately because of incompatibility in the measurement scales. These distinct measurement scales (counts versus cover) also reflect differences in animal assemblages. Percent cover is associated with populous sessile organisms such as barnacles and mussels, which form densely packed communities on rocky substrates. In contrast, motile invertebrates such as periwinkles and hermit crabs are much easier to individually enumerate in the field.

As with the infaunal analysis, epibiotic recovery assessments were based on interannual data collected at three core sites from each of three categories (Table 3). Data from the July surveys were used except for 1989 when the September survey was used in the analysis of upper transects. Upper-level transects were not sampled in July 1989 and shoreline treatment was not completed until September 1989 at two of the middle-intertidal core sites. The use of September 1989 data did not appear to introduce a seasonal bias because upper-level algal cover at control sites was consistent between September 1989 and the July surveys in subsequent years (Figure 24a). A few instances of missing (unsampled) epibiotic data in other years were scattered among Category-1 and 3 core sites. As with infauna, these multiple missing values were estimated by applying iterative techniques to the missing-value algorithms commonly used in randomized-

block designs (two-way ANOVAs). Application of the same missing value algorithm used in two-way ANOVAs was appropriate here because observations were identified by two classifications, treatment category and year. Estimation of missing values prior to testing parallelism was necessary to reduce temporal bias in the database as described in Appendix A.2.

In contrast to the infaunal analyses, inclusion of missing values was particularly important in tests of epibiota for parallelism. Without missing-value estimation, significant temporal bias could be introduced into the time profiles of category means. This is evident from the missing-value results for total algal cover shown in Figure 24 and Figure 25. Throughout monitoring, average cover along an individual transect tended to be consistently higher or lower than that of other sites within a category. For example, within upper transects at Category-3 sites (Figure 25a), the highest algal cover consistently occurred at Block Island. It also had missing observations in 1989 and 1990 that were estimated prior to computing the Category-3 mean shown as the thick solid line. The 1989 and 1990 category mean would have been much lower without these estimated values and an artificial increase would be introduced into the time profile between 1990 and 1991. Upon comparison with the Category-1 mean (Figure 24a), this artificial increase resulted in a significant departure from parallelism, which would be misinterpreted as a substantial repopulation event at Category-3 sites. In reality, this apparent recovery was an artifact of missing cover estimates at a site with naturally high algal cover.

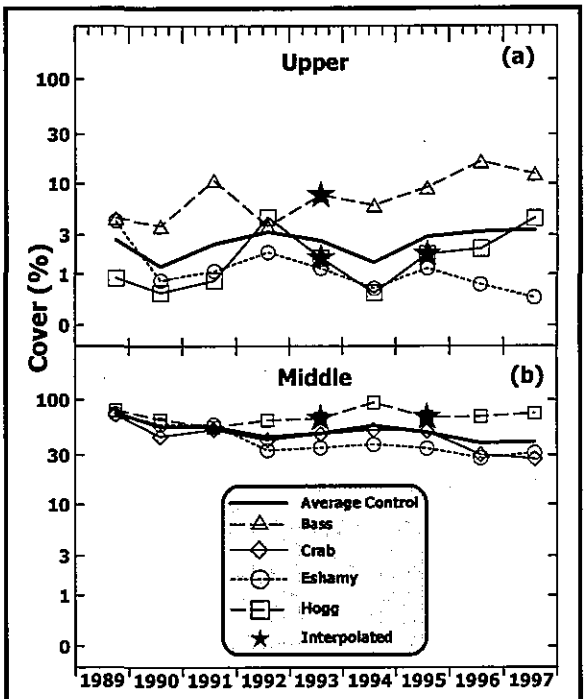


Figure 24. Algal cover at control (Category-1) sites along (a) upper and (b) middle transects

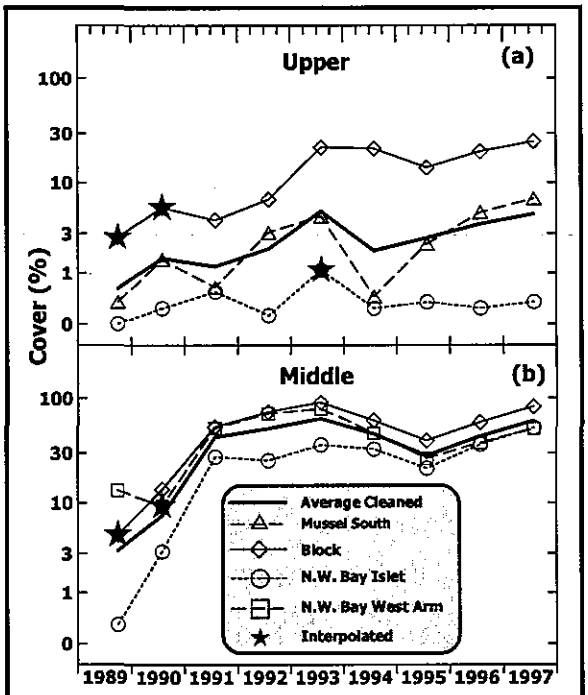


Figure 25. Algal cover at Category-3 (washed) sites along (a) upper and (b) middle transects

Results

Oil Cover

Variation in epibiotic hydrocarbon exposure is reflected by site-specific differences in the coverage and persistence of oil deposits. Differences among sampling sites and sampling elevations are portrayed in Figure 26. These data estimate the amount of oil coverage within the intertidal quadrats used for epibiotic enumeration. As such, they do not provide a comprehensive measure of oiling that occurred at a particular site. Nevertheless, they provide an index of hydrocarbon exposure experienced by the epibiota analyzed in this report.

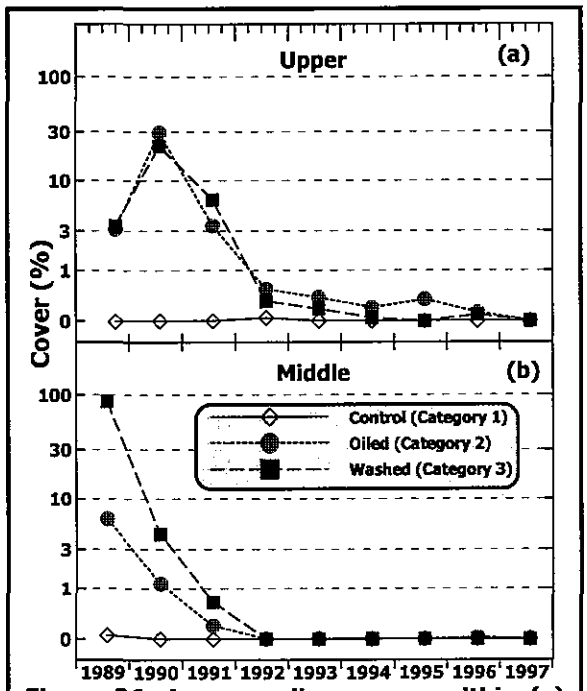


Figure 26. Average oil coverage within (a) upper and (b) middle transects

The greatest initial amount of oil cover was deposited along middle-intertidal transects at Category-3 sites. A measurable (>1%) amount of oil persisted along middle-intertidal elevations through 1990 at both Category-2 and 3 sites (Figure 26b). Although initially lower, tangible amounts of oil persisted for a year longer along upper transects (Figure 26a). Also, trace oiling (<0.5 %) remained evident for an additional four years at untreated sites. This contrasts with middle tidal transects where natural cleansing from wave action eliminated most surficial traces of oil by 1992.

Algae

Algal cover exhibited the same distinctive pattern of recovery that was observed in the infaunal time profiles, although there were small differences in the timing and magnitude of the repopulation event. Specifically, noncrustose algal cover within the middle-intertidal zone of impacted sites increased significantly after 1990. By 1992, algal cover had reached comparatively stable levels similar to those of control sites. This stabilized period coincided with an absence of measurable oil cover in the middle-intertidal zone (Figure 26b). Essentially all of the high rate ofalgal recovery was due to increases in rockweed (*Fucus gardneri*) cover. All other algal taxa were appreciably less abundant, smaller in size, and found beneath the *Fucus* frond canopy.

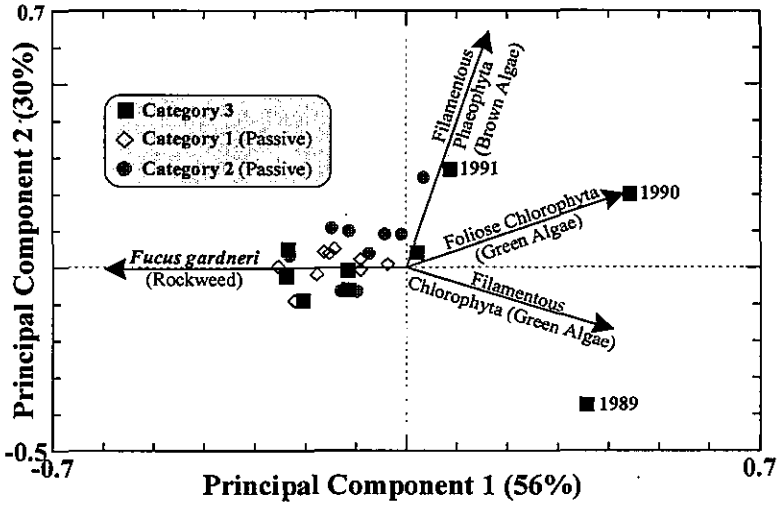


Figure 27. Principal middle-intertidal algal components at Category-3 sites

The importance of *Fucus gardneri* in overall algal recovery is revealed in a multivariate analysis of Category-3 algal cover (Figure 27). The interpretation of this principal component biplot follows infaunal ordinations shown in Figure 16 and Figure 20. Because the focus is on the strongest signature of recovery, the algal ordination is based only on Category-3 measurements (shown by ■ symbols). For comparison, the biplot shows the relative distribution of '*passive*' Category-1 (◇) and Category-2 (●) measurements, which do not participate in the determination of principal components. The first principal axis (x-axis) clearly discriminates between the algal community present immediately after the spill (1989 and 1990) and the algae present in recent years. In fact, this axis accounts for more than half (56%) of the variability among measurements.

The algal community present at Category-3 sites after 1991 was similar to that of control (Category-1) and Category-2 sites. This is indicated by the tight cluster of measurements with negative x-axis coordinates. These measurements are distinguished by the increased prevalence of *Fucus gardneri*, as disclosed by the biplot arrow for that taxon extending along the negative x-axis. The biplot arrows point in the direction of measurements that contain above-average abundance of the specified taxon. Measurements located in an opposite direction of a biplot arrow are deficient in that taxon. Thus, Category-3 sites were comparatively devoid of *Fucus gardneri* in 1989 and 1990. Although other algal forms were present in these early measurements (as shown by the other biplot arrows), a large part (34%) of the x-axis separation arose from differences in *Fucus* cover. Consequently, the parallelism tests described below were performed on *Fucus* cover alone.

Nevertheless, the succession of algal forms revealed by the biplot arrows is noteworthy. Trueblood *et al.* (1994) used the same ordination technique to determine directional changes in benthic infaunal communities through time. Their biplot contained a sequence of measurements that described a trajectory in ordination space. Each successional stage was characterized by a distinct taxonomic assemblage that was reflected by biplot arrows that pointed in the direction of measurements from a particular time. In a similar manner, Figure 27 can be interpreted in terms of algal succession. Immediately after the spill in 1989, filamentous green algae (chlorophyta) were numerically dominant forms. In the following year (1990) foliose forms became increasingly prevalent. Next, during the transition to the stable *Fucus* community (1991), the coverage of brown algae (Phaeophyta) increased within Category-3 quadrats. The precise mechanism for this early sequence of algal forms is a subject for further study. Attention now returns to the focus of this report, namely quantification of the timing and amplitude of the intertidal recovery process as a whole.

The initial impact to rockweed cover (*Fucus gardneri*) and its subsequent recovery are evident in time profiles at both Category-2 and Category-3 sites (Figure 28). As with infauna, the amplitude of the repopulation event is much larger at washed (Category-3) sites suggesting that the aggressive cleanup techniques had a greater initial impact on algae. However, after about 1991, both categories of impacted sites returned to relatively stable levels comparable to those of control sites (Category 1).

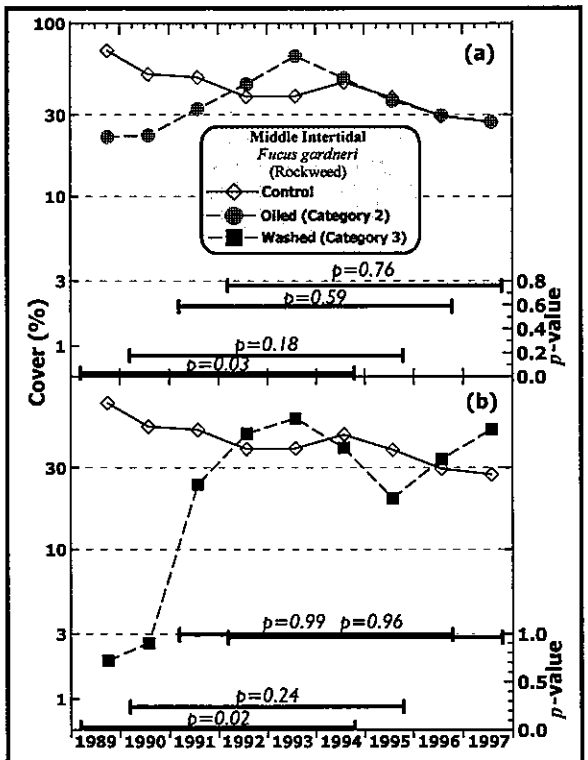


Figure 28. Parallelism tests on *Fucus gardneri* (rockweed) cover for (a) Category-2 and (b) Category-3 sites

Parallelism tests confirm this sequence of events. In particular, middle-intertidal *Fucus* cover at both Category-2 and 3 sites showed statistically significant ($p \leq 0.03$) departures from the temporal trends at control sites within the first six-year test window. This initial period, from 1989 through 1994, encompassed both years when *Fucus* was nearly absent (<4%) at Category-3 sites, as well as the abrupt increase to levels where it covered more than half of the middle-intertidal transects. Conversely, very high *p*-values (≥0.76) were observed in the last test window (1992-1997) indicating that recovery had been effectively

achieved. The decline in *Fucus* cover between 1993 and 1995 at Category-2 and 3 sites did not result in a statistically significant departure from parallelism. Nevertheless, the decline is evident at all three Category-3 sites (Figure 25b). It has been ascribed to the senescence (aging) of a single cohort (year-class) of *Fucus* recruited during the period of abrupt recovery in 1991 (Houghton *et al.*, 1997a).

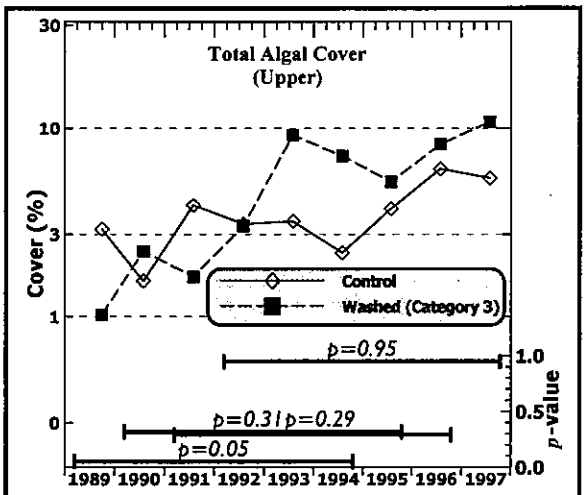


Figure 29. Parallelism tests on Category-3 algal cover along upper-intertidal transects

Algal cover within the upper-intertidal transects was an order of magnitude smaller than along middle transects (Figure 24). Despite this lower coverage, both *Fucus* and total algal cover exhibited statistically significant ($p \leq 0.09$) nonparallelism between control and impact time profiles within the first 6-year time window (cf. Figure 29).

Parallelism tests within subsequent windows disclosed high p-levels indicating that recovery had been achieved, albeit with a low (<10%) amount of cover. In addition, the repopulation event appeared to last longer at the upper transects with stabilization delayed until 1993 as opposed to 1991 or 1992 for the middle-intertidal transects. This apparent delay in recovery at the upper sites could have been due to the continued presence of residual oil along the upper transects (Figure 26). More likely, it is an artifact of the increased variability in upper-level cover estimates (*e.g.*, Figure 25). Further insight into the role of relict hydrocarbon contamination in delaying the recovery process along upper transects could be gained from canonical correspondence analyses between algal and oil coverage within each quadrat. However, site-specific environmental parameters such as quadrat elevation, slope, rugosity, exposure, and fetch should also be included in the CCA to avoid confounding anthropogenic impacts with natural variability.

Sessile Invertebrate Cover

There was no statistically significant evidence of impacts or recovery of sessile epibiotic invertebrates (barnacles and mussels). Namely, the null hypothesis that temporal trends in sessile invertebrate cover at impacted and control sites are parallel, could not be rejected in the initial 6-year time window. This was true of both upper and middle intertidal zones and at both Category-2 and 3 sites. This finding departs from that of algal cover where the first 6-year window exhibited significant levels of repopulation. However, differences in response to oiling and cleanup are

expected because sessile invertebrates, which consist mostly of acorn barnacles (*Chthamalus dalli* and *Balanus*), other barnacles (*Semibalanus*), and mussels (*Mytilus*), differs greatly from that of algae.

Although the departure from parallelism was not statistically significant at the p=0.10 level, the time profile of invertebrate cover revealed some visual evidence of the characteristic transient repopulation event that was seen in other intertidal assemblages. Specifically, the time profile of Category-3 invertebrate cover sharply increased between 1989 and 1991 (Figure 30b).

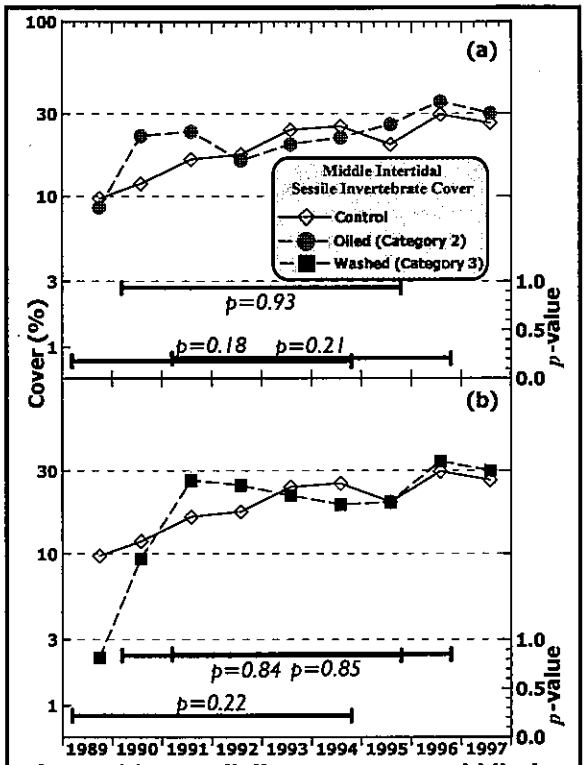


Figure 30. Parallelism tests on middle-intertidal invertebrate cover at (a) Category-2 and (b) Category-3 sites

The inability of the parallelism test to detect this trend was due to the earlier (1989) onset of intertidal repopulation that was contrasted against a slowly increasing trend in control populations. In comparison, the increase in rockweed coverage at washed sites was contrasted against declining control populations (Figure 28). In addition, attached invertebrates, as a single assemblage, occupied less primary space within the middle and upper intertidal zones than algal taxa. This increased the coefficient of variation in cover estimates.

In contrast to Category-3 sites, there was no strong visual evidence of recolonization by sessile invertebrates at Category-2 sites (Figure 30a) where less aggressive, or no oil-spill cleanup procedures were applied. This suggests that hot-water washes were substantially more detrimental to sessile invertebrates than hydrocarbon contamination alone. However, this result was inconclusive for reasons beyond the lack of statistically significance in Category-3 repopulation. Most of the sessile invertebrates reside in calcareous shells that make it difficult to determine whether the organisms are dead or alive, particularly when they are covered by oil. This could have resulted in overestimates of live invertebrate cover at Category-2 sites prior to 1992.

Motile Invertebrate Abundance

In contrast to sessile invertebrates, the abundance of mobile invertebrates exhibited a statistically significant departure from parallelism in the first 6-year time window at both Category-2 and 3

sites (Figure 31). Recovery of these middle-intertidal organisms was largely complete by 1991 as indicated by the high p levels (> 0.10) found in parallelism tests applied to subsequent time windows. The null hypothesis of parallelism could not be rejected for any of the time windows at the upper transects. Nevertheless, the upper-level Category-3 invertebrates displayed a transient early repopulation event characteristic of the recovery process.

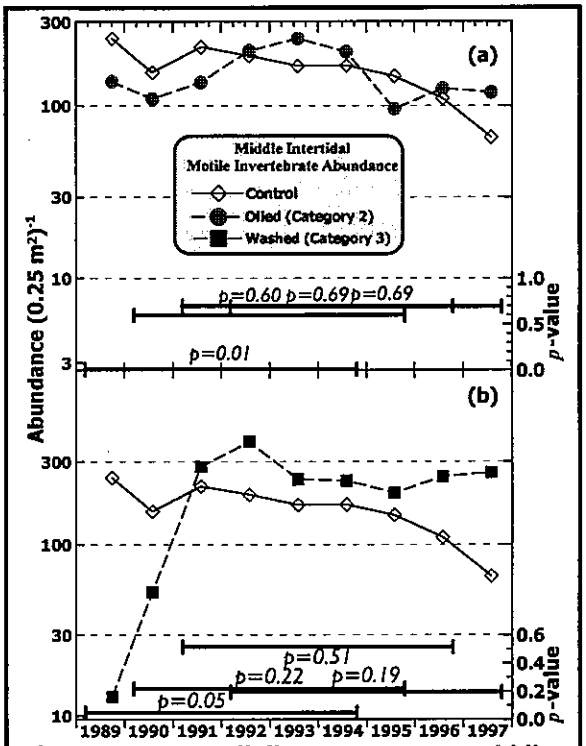


Figure 31. Parallelism tests on middle-intertidal invertebrate abundance at (a) Category-2 and (b) Category-3 sites

As with infauna, recolonization by several different epibiotic species contributed to the observed recovery in motile invertebrate abundance at middle-intertidal elevations. Enumerated invertebrates consisted mainly of motile grazer species with littorine periwinkle snails (*Littorina scutulata* and *L. sitkana*) being the most abundant. Limpets (Lottiidae) were another grazing invertebrate, while predatory drills (*Nucella*) and scavenger hermit crabs (*Pagurus*) were other common groups. Despite their dominance, only some of these taxa exhibited significant repopulation and those that did, recovered at different times and at different recolonization rates. In addition, post-colonization abundance at impacted sites was not always identical to that of the controls. Again, equality in absolute abundance is not necessarily expected due to the lack of randomization. In fact, total invertebrate abundance at Category-3 sites recovered to a level that consistently exceeded that of mean control sites (Figure 31b). This suggests that Category-3 sites may have supported a larger invertebrate population prior to the spill.

As with other intertidal assemblages, a principal component analysis identified the motile invertebrates that were instrumental in recolonizing the middle-intertidal zone at Category-3 sites (Figure 33). The interpretation of this biplot follows that of Figure 16, Figure 20, and Figure 27. Namely, the ordination is based only on Category-3 measurements (shown by ■ symbols) but includes the relative distribution of '*passive*' Category-1 (◇) and Category-2 (●) measurements, which do not participate in the determination of principal components. The first principal axis (x-axis) isolated the motile invertebrate community measurements at Category-3 sites immediately after the spill in 1989. The invertebrate community in subsequent measurements was much more

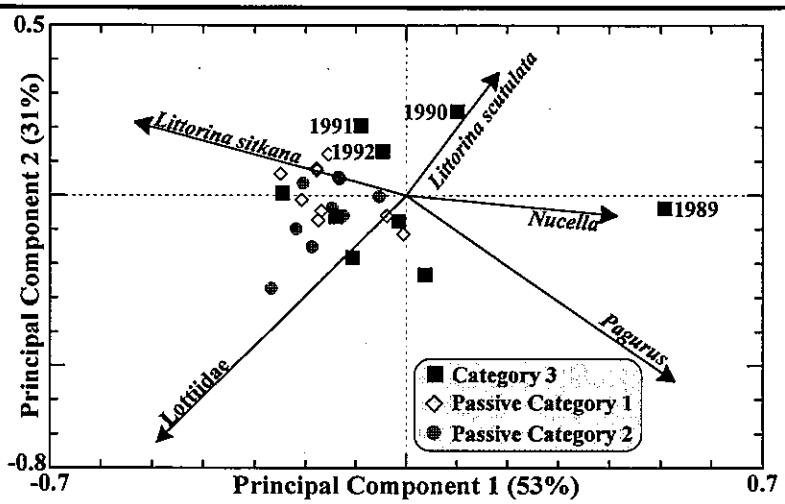


Figure 33. Principal components of invertebrate abundance at middle-intertidal Category-3 sites

consistent. Moreover, Category-1 and Category-2 measurements had comparable distributions of these Category-3 taxa. This is evident from the cluster of measurements with negative or near-zero x-axis components. The separation along this axis accounts for more than half (53%) of the variability among Category- 3 measurements.

As in previous biplots, arrows identify the taxa most responsible for the dissimilarity among measurements. Arrows pointing toward negative x-axis components identify invertebrate taxa that were relatively infrequent in the 1989 measurements. Thus, limpets (Lottiidae) and the Sitka periwinkle (*L. sitkana*) were responsible for most of the temporal differences among measurements. This is confirmed with parallelism tests (Figure 32). Relative to controls, both taxa experienced a sharp repopulation event prior to 1992. In the case of limpets (Lottiidae), abundance at Category-3 sites continued to gradually increase in the presence of declining control populations. Consequently, the parallelism hypothesis was rejected ($p \leq 0.004$) within all 6-year test windows (Figure 32a). Rapid early repopulation in both of these taxa was also the major contributor to the observed nonparallelism in motile invertebrate

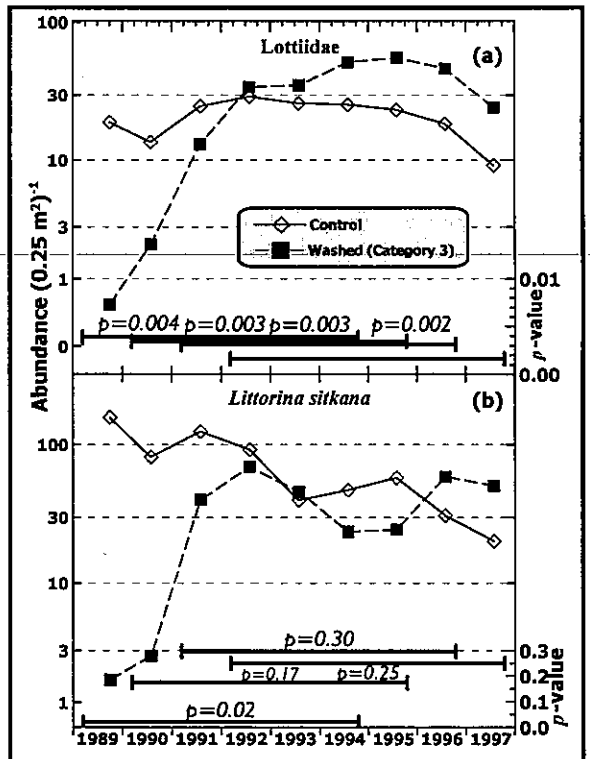


Figure 32. Parallelism tests on middle-intertidal abundance of (a) limpets (Lottiidae) and (b) the Sitka periwinkle (*Littorina sitkana*)

abundance within middle-intertidal measurements at Category-2 sites (Figure 31a).

Although there were no statistically significant departures from parallelism in total invertebrate abundance along upper transects, limpets (Lottiidae) and periwinkles (*Littorina*) again exhibited an abrupt repopulation signature characteristic of the recovery process (Figure 34). However, along the upper transects the checkered (*L. scutulata*) rather than the Sitka (*L. sitkana*) periwinkle exhibited strong early repopulation. The Sitka periwinkle (*L. sitkana*) is generally known to be less prevalent within the upper tidal reaches. This could result from differences in reproductive characteristics. In contrast to the pelagic larval stage of checkered (*L. scutulata*) periwinkle, the Sitka periwinkle (*L. sitkana*) is a brooder that lays benthic egg masses on, or under algae (Behrens-Yamada, 1989). However, wave impact and desiccation may also play a role in the distribution of the Sitka periwinkle (Behrens, 1972).

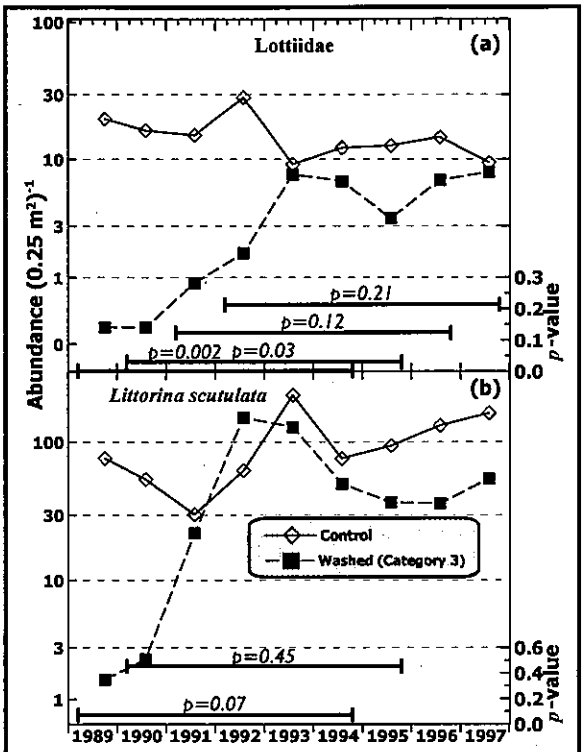


Figure 34. Parallelism tests on upper-intertidal abundance of (a) limpets (Lottiidae) and (b) the checkered periwinkle (*Littorina scutulata*)

Limpet (Lottiidae) abundance within upper transects at impacted sites (Figure 34a) did not continue to increase relative to controls, as was the case along middle transects (Figure 32a). This is reflected in the inability to reject the hypothesis of parallelism ($p>0.10$) within the last two test windows at upper-levels. In addition, the onset of limpet recolonization occurred later along upper transects. The delayed recolonization may be a consequence of a prolonged exposure to trace hydrocarbon contamination within the upper tidal reaches (Figure 26).

Recommendations

This chapter provides strong inferential evidence that all major epibiotic assemblages experienced marked repopulation at impacted sites before 1993. The recolonization event was similar in timing and magnitude to the infaunal recovery described in the previous chapter. Subsequently, the populations of all the impacted intertidal assemblages stabilized and began tracking control populations, at least within the statistical resolution of the data collected to date. As with

infauna, future monitoring efforts should emphasize resolution of subtle ancillary trends in the recovery process. The following recommendations are made with that in mind.

- Continue epifaunal monitoring at the eighteen upper and middle transects at the eleven core sites list in Table 3. As with infaunal sampling, if the field sampling effort is limited during future surveys, collection of samples from these eleven sites should take precedence over sampling at other epibiotic sites. These sites have been consistently sampled on an annual basis throughout the program, including the years during major epibiotic recolonization between 1990 and 1992.
- If resources are available, conduct monitoring at additional control sites. This would better define natural population fluctuations that result from regional influences within Prince William Sound. This, along with an improved estimate of inherent spatial variability among sites, would further enhance the ability to detect shifts in impacted epibiotic communities.
- Increase the number of replicate quadrats within upper transects. In the current database, upper transects contain half the number of quadrats measured within the middle intertidal zone. At the same time, patterns of impact and recovery are inherently more difficult to quantify along upper-elevation transects. The upper-elevation zone represents the highest elevation range of species distributions. Consequently, it consists of mostly barren rock with only patchy occurrences of epibiota. In addition, elevation differences along the upper-intertidal transects introduces variability. Upper-elevation transects at some sites were established at elevations equivalent to the mid-intertidal transects at other sites. Both of these aspects of upper-intertidal data induce larger spatial and temporal variability as reflected in a comparison of the top (a) and bottom (b) frames in Figure 24 and Figure 25. Because of the presence of more substantial within-site spatial variation, this zone requires more extensive sampling to discern subtle temporal trends.
- Conduct canonical correspondence analyses to distinguish between within-site environmental variability and various anthropogenic impacts in the database. Assessment of the influence of small-scale variability in physicochemical properties on fauna will lend further insight into the role of various cleanup strategies. For example, these analyses would quantify the role of relict hydrocarbon contamination in delaying the algal recovery along upper transects. To accomplish this, variation in site-specific environmental parameters, such as quadrat elevation, slope, rugosity, exposure, and fetch, would have to be incorporated.
- Apply population models to shed light on post-recolonization fluctuations observed in the time profiles at impacted sites. The post-recolonization population fluctuations have a distinct character that reverberates the steep initial increase in abundance. Immediately after the large initial increase in population, abundance continues to gradually increase for a few years relative to control populations. Subsequently they slowly decline to a local minimum and then begin increasing again. The amplitude of the fluctuations is much smaller than the initial population increase and cannot be resolved with parallelism tests. Houghton *et al.* (1997a) ascribes the post-recolonization fluctuations in rockweed to the senescence (aging) of a single cohort (year-class) of germlings recruited during the major repopulation event of 1991-1992. Although rockweed (Figure 28b) exhibits the clearest signature, a wide range of taxa reflect a similar trend (*cf.* Figures 11-13, 17, 18, 21, 28a, 29-31, 33, 34). The exponential

population model (Equation 3) used in the parallelism tests is incapable of delineating these fluctuations. However, a slightly more-general model that accounts for resource limitations can represent time profiles that stabilize about the carrying capacity of the environment (Verhulst, 1838). In addition, complex population dynamics can be obtained by formulating this simple logistic population model for discrete time measurements (Ricker, 1975). Where reproduction is rapid, the model predicts damped oscillations that are highly reminiscent of the observed fluctuations. These population models can be applied without explicit age-structure information and can be used to determine the carrying capacity (equilibrium abundance) and the intrinsic rate of population increase for individual taxa (Ives, 1995). More importantly, the model will predict when reverberations from the initial recolonization become imperceptible relative to ambient fluctuations. At that point, populations can be said to have fully stabilized relative to control populations.

Chapter 4. Conclusions

The analyses described in this report quantitatively resolve a number of important questions surrounding long-term recovery of intertidal biota after the *Exxon Valdez* oil spill. Results that pertain to some of these questions are summarized below.

Have the intertidal biota of Prince William Sound recovered?

Yes, this report provides strong inferential evidence that intertidal populations within Prince William Sound experienced a substantial amount of recovery from the effects of the 1989 oil spill and cleanup. The onset of recovery, as defined by a sharp increase in abundance at impacted sites relative to controls, began less than three years after the spill. Recolonization required about one to two years and populations stabilized for most taxa by 1993. During this recolonization, populations increased by a factor of eight on average. The repopulation was large enough to be detected by statistical tests. The tests quantified temporal parallelism between the abundance time profiles at impacted and control sites within a moving 6-year time window. For most major intertidal assemblages, there was a statistically significant departure from parallelism within the initial 6-year window indicating that a significant repopulation had occurred at impacted sties relative to control sites. In contrast, time windows that spanned subsequent years showed a high-degree of parallelism indicating that the impacted populations had stabilized and had begun to more-closely track fluctuations in control populations.

The magnitude and scope of this abrupt repopulation event provided compelling evidence that intertidal populations had materially recovered by 1993. Statistically significant repopulation was evident throughout the intertidal zone at both Category-2 (oiled) sites and Category-3 (hot-water washed) sites. Widely disparate intertidal assemblages, including infauna, algae, and epifaunal invertebrates, began recolonizing at about the same time (1990-1991) and over a similar duration (one-to-two years). While smaller-amplitude perturbations in community structure and abundance may still be occurring, most of the recovery from the spill took place during recolonization prior to 1993. Subtle trends within impacted populations are visually evident in time profiles after 1993, but they cannot be currently resolved with statistical hypothesis tests based on parallelism with control populations. In addition, average infaunal abundance at Category-3 sites stabilized at a level well below that of the control sites. Whether this was simply an artifact of nonrandomization or if impacted infauna have yet to return to pre-spill conditions cannot be determined with the current database.

How did the timing of recovery differ among the various intertidal assemblages and organisms?

Overall, the timing of the recruitment event was remarkably similar across the broad range of intertidal flora and fauna. Nevertheless, individual taxa exhibited measurable differences in the onset and duration of the recolonization. Table 4 summarizes these differences for the taxa examined in this report. The statistical significance of the recruitment event is indicated in the first column. It reports the p-value for the parallelism test performed on the first six-years of data. Small values ($p<0.10$) are shaded and indicate a significant departure from parallelism between control and impact time profiles. In those cases, the repopulation was large enough to be detected with some degree of statistical confidence. However, the test applies a linear functional form (Equations 1 and 2) across a 6-year span. Consequently, it is better suited for evaluating gradual (linear) trends over longer (6-year) periods, rather than the abrupt step-function reflected in many of the time profiles. Thus, higher p-values do not necessarily mean that substantial repopulation did not occur; they simply indicate that it was not detected by the statistical construct. Nevertheless, the tests provide a rigorous quantitative method for comparing fluctuations in control and impact populations. Taxa whose time profiles at impacted sites exhibited marked repopulation before 1993 are listed regardless of their p-value. Most of these revealed a statistically significant departure from parallelism (ongoing recovery) in the first 6-year test window but not in subsequent test windows (recovery complete). Individual species that exhibited significant recolonization were often not the dominant organisms.

The timing of repopulation at impacted sites was determined graphically using the logarithmic time profiles shown throughout this report. The onset of recolonization occurred within the year following the date listed in Table 4. It represents the last survey when abundance was markedly below that of subsequent surveys. Most intertidal organisms began recovering between 1990 and 1992, although epifaunal invertebrates were early colonizers of Category-3 sites. Their populations began increasing between 1989 and 1990. The recovery of some individual infaunal taxa, such as the tube-building worm *Laphania boecki* and the small bivalve *Rochefortia tumida*, was somewhat delayed although their overall abundance was comparatively small (Figure 17 and Figure 22). For most taxa, sharp increases in population levels ended by 1992, although algal cover in the upper-intertidal zone continued to increase into 1993. The extended recovery within this zone may have resulted from prolonged exposure to tangible amounts of oil that persisted along upper transects (Figure 26a). However, quantitative determination requires the application of a canonical correspondence analysis that incorporates other site-specific environmental parameters, such as quadrat elevation, slope, rugosity, exposure, and fetch.

Table 4. Timing and magnitude of recovery

Category / Elevation / Assemblage	Parallelism [a]	Repopulation: Onset [b]	Stabilization [c]	Amplitude [d]
Category 2				
Lower Intertidal				
Total Infaunal Abundance	0.194	1991	1992	2.4
Annelida	0.236	1991	1992	2.7
Other Infaunal Taxa [e]	0.363	1991	1992	2.3
Mollusca	0.051	1991	1992	2.1
Fabriciola berkeleyi	0.070	1991	1992	7.1
Laphania boecki	0.044 [f]	1993	1994	3.6
Middle Intertidal				
Total Algal Cover	0.054	1990	1991	1.8
Fucus gardneri Cover	0.027	1990	1991	1.7
Total Invertebrate Abundance	0.010	1990	1992	1.8
Littorina sitkana	0.011	1991	1992	3.6
Upper Intertidal				
Total Algal Cover	0.056	1991	1993	1.8
Fucus gardneri Cover	0.092	1991	1993	1.6
Total Invertebrate Abundance	0.279	1991	1992	2.3
Category 3				
Lower Intertidal				
Total Infaunal Abundance	0.046	1991	1992	4.1
Annelida	0.281	1991	1992	5.4
Other Infaunal Taxa	0.231	1990	1992	7.1
Mollusca	0.005	1991	1992	6.7
Cumella vulgaris	0.001 [f]	1991	1992	3.6
Eteone longa	0.027 [f]	1991	1992	3.1
Fartulum occidentale	0.006	1991	1994	11.0
Rochefortia tumida	0.151	1993	1995	5.1
Saccocirrus eroticus	0.391	1991	1992	6.8
Syllis alternata	0.089	1991	1994	2.8
Middle Intertidal				
Total Algal Cover	0.035	1990	1991	8.5
Fucus gardneri Cover	0.023	1990	1991	13.3
Total Invertebrate Cover	0.218	1989	1991	8.4
Total Invertebrate Abundance	0.047	1989	1991	19.1
Limpets (Lottiidae)	0.004 [f]	1989	1992	40.3
Littorina sitkana	0.018	1990	1991	16.0
Upper Intertidal				
Total Algal Cover	0.049	1991	1993	2.6
Fucus gardneri Cover	0.013	1991	1993	3.9
Total Invertebrate Abundance	0.139	1989	1991	38.4
Limpets (Lottiidae)	0.002	1990	1993	10.4
Littorina scutulata	0.070	1990	1991	22.8

[a] Statistical significance of parallelism test performed on first 6-year test window. Cells are shaded when the null hypothesis of parallelism between time profiles at impacted and control sites is rejected ($p \leq 0.10$), suggesting that organisms were measurably impacted and that their recovery was in progress during the first six years of monitoring. Unless otherwise indicated, recovery was eventually achieved because parallelism could not be rejected at the stated significance level within one or more subsequent test windows.

[b] Graphical estimate of the year when the abrupt repopulation event began

[c] Graphical estimate of the year when populations stabilized after the steep repopulation period

[d] Graphical estimate of the factor of increase from impacted population levels

[e] Combined abundance of infaunal taxa other than annelids, mollusks, and crustaceans

[f] Null hypothesis of parallelism was also rejected across all subsequent 6-year test windows

How did recovery differ at sites that were subjected to high-pressure washing with hot water?

The repopulation amplitude listed in Table 4 represents the ratio between stabilized and impacted populations. It is clear from the Table that the increase in intertidal populations was much larger at sites that were subjected to high-pressure hot-water washes (Category-3 sites). At Category-3 sites, the average population increase was a factor of eleven, while Category-2 populations only increased by a factor of three on average. Each individual assemblage and taxon listed under both impact categories had a consistently larger population increase at Category-3 sites. These differences are illustrated in Figures 1, 5, 11-13, 28, 30, and 31.

The most logical explanation for these differences is that damage to intertidal biota was more severe at sites that were cleaned by high-pressure hot-water washes. However, the enhanced recovery at these sites compensated, at least in part, for the increased damage. The increased biological damage from high-pressure hot-water washes has been documented for algae (van Tamelen and Stekoll, 1996), mussels (Lees *et al.*, 1996), predatory snails (Ebert and Lees, 1996), infauna (Driskell *et al.*, 1996), epibiota (Houghton *et al.*, 1996), and marine ecosystems in general (Mearns, 1996). However, many of these studies suggested that recovery at treated sites was slower or incomplete. In contrast, the results of this investigation indicated that long-term intertidal recovery, as measured by abundance at impacted sites, occurred at about the same time.

If intertidal populations have recovered, why should monitoring continue?

There are a number of important remaining issues surrounding the recovery of intertidal biota. These issues can only be resolved with continued monitoring and analysis. By necessity, quantitative assessment of recovery was based on statistical tests of parallelism between control and impact populations. Without sufficient pre-spill data, tests based on convergence in absolute population levels were untenable. Infaunal populations at washed (Category-3) sites were lower than at other sites, and these differences were related to a deficiency in fine sediments at Category-3 sites. Continued monitoring with manipulative experiments may further elucidate the relationship between environmental factors and intertidal organisms. Experiments can also be conducted that indicate whether hot-water washing preferentially winnowed fine sediments at Category-3 sites.

In addition, the tests applied in this report used only the most fundamental community measure; namely, the average abundance of major assemblages. Although total abundance may have stabilized, shifts in the community composition within the major assemblages may be indicative of subtler aspects of recovery. Related to this, the duration of the current database limits the ability

of parallelism tests to detect ongoing fluctuations in abundance. Continued monitoring will increase the test's ability to resolve gradual long-term trends not currently evident in the data. While changes in abundance as large as the major recolonization prior to 1993 are not expected, continued monitoring would confirm whether there are any perceptible lingering effects from the oil spill.

There is some indication of these lingering effects in the current database. The population of some individual taxa took longer to stabilize (Table 4) and others, such as little neck clams (*Protothaca staminea*), may be still in the process of recovering (Shigenaka *et al.*, 1999). In addition, reverberations from the abrupt recolonization event are apparent in many time profiles. Houghton *et al.* (1996) ascribed an oscillation in rockweed cover to the senescence of a single cohort (age-class) that colonized during a brief one-year period between 1990 and 1991 (Figure 28b). However, many other taxa, such as infaunal mollusks (Figure 15), exhibit a similar post-recolonization trend. Application of population models to the current database would determine when oscillations are likely to decay to imperceptible levels. Continued monitoring would refine and confirm these predictions of long-term recovery. Thus, the parallelism tests use empirical data to characterize the past while population models predict when all repercussions from the oil spill will disappear from the intertidal communities of Prince William Sound.

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A.1 Glossary of Selected Ordination and Statistical Terms

Alternative Hypothesis — H_a: The oil spill affected the abundance of intertidal organisms, and temporal profiles of abundance at control and impact sites are not parallel.

ANOVA — An acronym for analysis of variance that examines the contribution of each parameter to the variation in the outcomes of an experiment. It is a method of statistical analysis broadly applicable to a number of research designs, used to determine differences among the means of two or more groups on a variable.

β-Diversity — Beta diversity measures the differences in diversity among samples. A group of samples with high β-diversity will have completely different species compositions and some pairs of samples may have no species in common. A group of samples with low β-diversity will be similar in species composition throughout. Principal Component Analysis (PCA) functions best at low β-diversity while correspondence analysis behaves best at high β-diversity (ter Braak, 1983).

Biplot — An ordination diagram which simultaneously displays both the distribution of samples in ordination space as well as the taxa most responsible for separating those samples. Samples are represented by points whose proximity is an indication of the degree of similarity between the intertidal communities within those samples. Biplots have specific rules for interpretation as described by Gabriel (1971). Species are represented by a biplot arrow that points in the direction of maximum change in abundance. The length of the arrow is proportional to maximum change in the abundance of that particular species (and its importance in the ordination). The relative abundance of a species within samples can be inferred by projecting the sample points onto that species' vector. Also, arrows that point in the same direction indicate that the species are closely correlated in samples whereas arrows oriented at right angles exhibit little or no relationship. Oblique angles indicate negative correlation.

Canonical Correspondence Analysis (CCA) — A combination of ordination and multiple regression where ordination axes are constrained to be linear combinations of environmental variables. Results are presented in the form of triplots where environmental variables are represented by vectors or arrows

Dispersion Measure — A measure of the dispersion of replicates or other observations around the central tendency (mean or median) of the distribution. Dispersion can be represented through the use of error bars or range limits.

CNESS — An acronym for the **C**hord-**N**ormalized **E**xpected **S**pecies **S**hared distance metric (Trueblood *et al.*, 1994). It is described in Appendix A-3.

Coefficient of Variation A coefficient used to compare the relative amounts of variation in populations having different means. It is defined as the standard deviation expressed as a percentage of the mean.

Correspondence Analysis An eigenanalysis-based ordination method also known as reciprocal averaging where sample scores and species scores are calculated simultaneously as weighted average of one another by maximizing the correlation between them. These methods perform best when species have unimodal distributions along environmental gradients.

Distance Metric A measure of the dissimilarity in species composition between two samples. Dissimilarity metrics are large for samples which share no species, and near zero for samples which have identical species composition. Metrics obey the triangular inequality rule which improves the behavior of multivariate analyses and geometric interpretation. CNESS is an example of a distance metric. The Bray-Curtis index is not a metric measure.

Horseshoe Effect A distortion of ordination diagrams that is evident as strong curvature in the distribution of sample scores in the first two principal axes. The curvature can be strong enough that scores along the first axis are involuted and form a horseshoe shape. The horseshoe effect is an artifact of ordination techniques, such as Principal Component Analysis, when they are applied to very long gradients where few species are shared between widely separated samples (high β-diversity). Correspondence analyses tend to reduce the severity of the horseshoe effect.

Least-Squares Regression The process of fitting a function (here a polynomial) to data (abundance versus time) such that the sum of the squared residuals is minimized.

Monte Carlo Permutation Test A direct method for evaluating the significance of ordination results by maintaining actual data values but randomly permuting the sample assignments. This results in nonparametric hypothesis tests that do not depend on distributional assumptions typical of commonly used statistical tests (*e.g.*, the student's t or F-tests).

Multicolinearity High correlation among a group of several (more than two) variables.

Multispecific A taxon that consists of more than one species.

Multivariate Analysis An analysis that simultaneously examines the behavior of more than one dependent variable, such as the abundance of many different species, as a function of independent variables (*viz.*, samples). Multivariate methods take advantage of joint structure among the species in the intertidal community by accounting for multicolinearity (multiple intercorrelations) in their response to external (environmental) factors.

Null Hypothesis H_0: The oil spill had no effect on the abundance of intertidal organisms, and temporal profiles of abundance at control and impact sites are parallel.

Ordination A multivariate technique that arranges or "orders" (as in ordination) samples along an axis based on species composition. This ordination can be conducted along a number of dimensions (usually 2 or 3) that approximate some pattern of response of the intertidal community to underlying environmental gradients (such as grain size). Thus, ordination condenses the complex species-abundance database to a few factors responsible for observed variability within the intertidal community, while retaining ecologically meaningful biological information.

Parallelism A condition where time profiles of average abundance at control and impact sites track one another through time. Observed temporal excursions must act in unison so that the average levels change at the same time while maintaining a constant difference in (logarithmic) abundance.

Passive Samples Passive samples (or passive species) can be added to any existing principal-component or correspondence analysis without affecting the original ordination. They are scaled using the eigenvectors and eigenvalues that were derived in the original ordination of "active" samples. As such, passive samples do not directly participate in the ordination but their relative position in the ordination space can be shown after the fact.

Principal Components Analysis An ordination technique that involves an eigenanalysis of the correlation matrix. Ideally, the first Principal Component will represent the dominant environmental gradient. The second Component will be orthogonal (completely uncorrelated) with the first, and will explain some of the residual variation. This class of ordination techniques works best for monotonic distributions where species abundance steadily increases or decreases along an environmental gradient. In reality, organism abundance tends to have a unimodal distribution but may appear to be monotonic if small portions of the gradient are sampled.

***p*-Value** The probability of rejecting the null hypothesis of no impacts (H_0) when the null hypothesis is true (Type I error). Commonly, *p*-values of 0.10, 0.05, or 0.01 are established as indicating statistical significance at the 90%, 95%, or 99% confidence level. In this study an observed *p*-value is considered statistically significant if it is below 0.10 which indicates a high degree of confidence that the time profiles are not parallel and recovery from impacts is ongoing.

Randomized Block Design A two-way ANOVA. In this oil recovery assessment, observations are identified by two classifications, treatment category and year.

Sample Score The coordinates along ordination axes specifying the location of a sample. They are often related to environmental gradients and represent the specific intertidal community that is best suited to an ecological niche.

Sequential Regression A sequence of regressions where additional polynomial terms are added to the model until additional terms do not contribute significantly to the overall fit. The additional higher-order terms are entered one at a time, until no more variables explain significant variation.

Time Profile Time Sequence Temporal sequence of annual abundance levels computed from the average of abundance over sites within a single treatment category.

Time Window Portion of the time profile over which the test for parallelism is applied. In this study, the time window spanned six years. After the test is applied in the first six-year time window, the time window is shifted so it starts one year later and the parallelism test is reapplied across the new (overlapping) window.

Transformation A mathematical operation performed on a variable (*viz.*, species abundance or environmental variables), with the goal of making that variable more biologically meaningful, conform to a statistical distribution (usually Gaussian), achieve additivity, or have stable variance (homoscedasticity).

Triplot In canonical analysis, three sets of scores (species scores, sample scores, and environmental variable scores) are plotted simultaneously. Sample scores and species scores are usually indicated by symbols or labeled points. Continuous environmental variables are indicated by arrows. As with biplots, these plots have specific rules for interpretation.

Type I Error Incorrectly rejecting the null hypothesis (H_0) that the oil spill had no effect on the abundance of intertidal organisms, when in fact there was no effect.

Type II Error Incorrectly accepting a false null hypothesis that the oil spill had no effect on the abundance of intertidal organisms, when in fact the alternative hypothesis (H_a) that the oil spill affected organism abundance is true.

Unimodal Distribution A species frequency distribution with one mode indicating that the species has one optimal environmental condition. Any increase or decrease in environmental conditions from this optimum will be less hospitable to the species and result in lower abundance. Ordination techniques based on correspondence analysis perform best when species have unimodal distributions.

A.2 Missing Values

In some cases, missing values can invalidate tests for parallelism unless replacement estimates are included in the computation of mean profiles prior to testing. As demonstrated below, missing values at one site can artificially introduce temporal trends in profiles of mean abundance that are deceptively similar to the abrupt recolonization observed at impacted sites. To reduce the influence of missing values, the parallelism tests described in this report incorporate an estimate of the missing values. This is analogous to missing value estimation commonly implemented in an ANOVA of a randomized block design (Section 11.7 of Sokal and Rohlf, 1997). As with ANOVAs, the parallelism tests presented in this report were not routinely conducted with and without estimated missing values. Instead, the validity of the statistical analyses depended on the incorporation of missing-value estimates. This is clear from a hypothetical example that closely resembles the circumstances within intertidal data from Prince William Sound.

Suppose the true abundance at three individual sites remains unchanged through time as shown in Figure A.1a. Under time-invariant environmental conditions, this would apply to intertidal assemblages at control sites or at impacted sites that were not affected by the oil spill and cleanup. Offset time profiles similar to this are typical of a number of assemblages, taxa, and species found in the intertidal data where absolute abundance among the sites is consistently different through time. Algal cover along upper transects (Figure 24a) and cover of the small acorn barnacle (*Chthamalus dalli*, Figure 3b) are two examples. Hogg Bay site provided a more hospitable environment for small acorn barnacles than did the other two control sites and the Bass Harbor site had a higher algal cover than the other control sites.

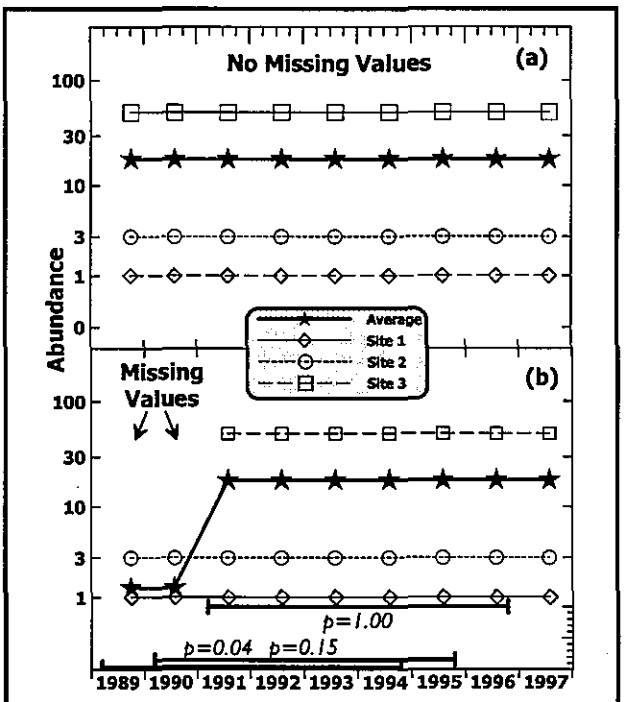

Figure A.1. Hypothetical time profiles (a) without missing values and (b) with missing values

If data had not been collected from the high-abundance site during the first two-years of sampling, then the computed mean profile would exhibit a sharp increase in abundance in between 1990 and 1991 (Figure A.1b). In the absence of other information, this apparent repopulation

could easily be misconstrued as ongoing recovery from impacts. If it were compared to a control group whose average time profile was similar to that of Figure A.1a, then the apparent impact and recovery would be confirmed with a statistically significant departure from parallelism immediately after the spill ($p<0.10$).

This example demonstrates the importance of including missing-value estimates before computing category means. Without inclusion of estimates, missing values can introduce a Type I error which results in false conclusions of a population impact and subsequent recovery. A similar dialectic can be constructed that would illustrate how missing values can also lead to Type II errors, where an impact and subsequent recovery is missed.

A.3 CHORD-NORMALIZED EXPECTED SPECIES SHARED

The CNESS distance metric was used in the principal component analyses conducted as part of this study. CNESS is an acronym for chord-normalized expected species shared and is one of many '*distance*' measures that succinctly quantify similarities or differences in biological communities between samples. It offers many advantages over other commonly used measures such as the Bray-Curtis similarity index. First, because CNESS is a true metric, it is not subject to instabilities during eigenanalysis. Consequently, ordinations can be strictly interpreted on a biplot. Second, the influence of rare versus dominant taxa can be quantitatively controlled by changing the subsample size (Grassle and Smith, 1976). Other distance measures that focus on dominant species (*e.g.*, Bray-Curtis) have been criticized for their lack of sensitivity to rare species (Peet, 1974; ter Braak, 1983).

This study applied CNESS in a principal component analysis similar to that of Trueblood *et al.* (1994). They first described a metric scaling of CNESS distances such that the species-sample matrix is converted to hypergeometric probabilities, normalized, and then centered. Principal component analysis of this matrix, denoted PCA-H, provides an ordination of CNESS distances among samples. They used PCA-H to quantify seasonal succession of infaunal organisms within Boston Harbor.

By adjusting the subsample size in CNESS, the influence of dominant organisms can be quantitatively controlled. If a very small subsample of m organisms is removed from the original sample of N organisms, then it is likely that it will consist of dominant organisms. Conversely, dominants have their least influence on CNESS when the subsample size m is equal to the sample size N; *i.e.*, all the organisms are included in the analysis. Thus, as larger numbers of organisms are drawn, it is more likely a rare species will be drawn in the subsample. In this report, an intermediate subsample size was selected so that both rare and dominant taxa participated in the ordination.

In practice, subsampling is performed analytically through application of hypergeometric probabilities, which measure the likelihood of selecting a particular species in a random draw of m organisms from the sample. Using hypergeometric probabilities, the expected number of species shared ($ESS_{ij|m}$) between subsamples of size (m), is given by:

$$ESS_{(ij|m)} = \sum_{k=1}^{S} \left\{ \left[1 - \frac{\binom{N_i - n_{ik}}{m}}{\binom{N_i}{m}} \right] \left[1 - \frac{\binom{N_j - n_{jk}}{m}}{\binom{N_j}{m}} \right] \right\}$$

where N_i is the total number of individuals in the i^{th} sample and n_{ik} is the abundance of the k^{th} species in the i^{th} sample. The CNESS distance between two samples i and j is given by:

$$\mathbf{CNESS}_{\langle ij|m\rangle} = \left\{ 2 - \left[\frac{2\,ESS_{\langle ij|m\rangle}}{\left(ESS_{\langle ii|m\rangle} ESS_{\langle jj|m\rangle}\right)^{1/2}} \right] \right\}^{1/2}$$

This formulation results in the CNESS family of distance measures that range between 0 and $\sqrt{2}$. Higher values represent greater dissimilarity between samples.

A.4 Time Profiles of Abundance with Standard Errors

The mean-abundance time profiles presented in the body of this report are reproduced in this appendix along with a measure of variance about individual category means. The standard error bars reflect the relative spread of site abundance measurements for a given category during a given year. Although these plots are provided in response to requests from reviewers, a cautionary note concerning their interpretation is worth repeating. The average abundance level and the distribution of site-specific abundance at any given time have little bearing on the determination of intertidal recovery. In particular, large error bars that overlap mean abundance at control and impacted sites do not imply that these communities have converged and that recovery has been achieved. This is the conclusion that would be reached using the definition of recovery advanced by Ganning *et al.* (1984) and applied in previous assessments of this intertidal database. Those analyses consider recovery complete when population variability at impacted sites is within the range of natural fluctuations at control sites.

However, control and impact sites within the Prince William Sound differ systematically in ways unrelated to oil exposure or cleanup technique. Also, large zoogeographic differences among the monitoring sites introduces substantial inherent variability about the category means computed for a particular year. Consequently, recovery assessments based on convergence in absolute community measures are statistically untenable. This is evident in the figures shown in this Appendix; namely, with three samples, 90% confidence intervals span about three times the interval shown by the standard error bars and often encompass all other category means. Consequently, within any given year, the difference in category means cannot be reliably discerned. This of course, does not imply that there were no intertidal impacts or that recovery had occurred prior to the implementation of monitoring. Instead, it indicates that some of the sites within a particular category were not as hospitable to certain intertidal taxa, which results in large geographic variability. Insofar as recovery, analysis of variance in mean abundance at any given time is not pertinent. Instead, recovery can be reliably detected when temporal changes in impacted populations begin to reflect fluctuations in control populations.

Nevertheless, within-year category means and the associated variance measures are of interest for reasons other than assessing recovery. For example, they reveal the degree of spatial variability among various intertidal assemblages. With that in mind, the following figures show standard errors on the mean of each within-year category means. To visually distinguish among overlapping limits, the survey times for individual category means are shifted slightly on the plots in this appendix. The captions cite the original Figure in the body of the report to which they correspond.

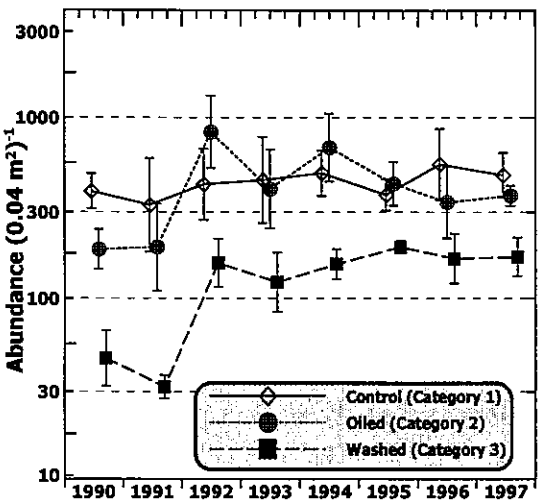

Figure A.2. Average infaunal abundance ± standard error (Figure 1)

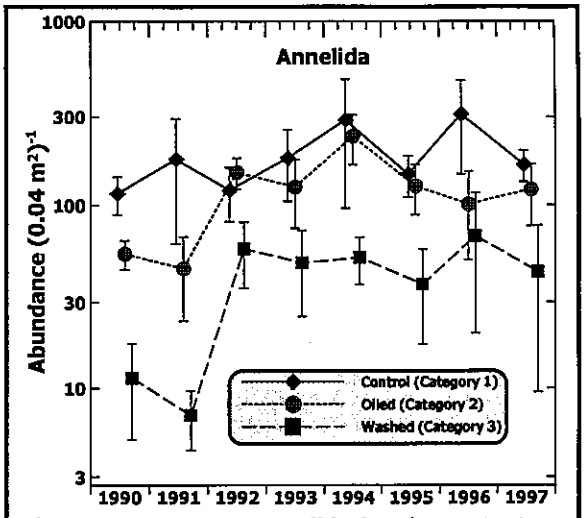

Figure A.3. Average annelid abundance ± standard error (Figure 11)

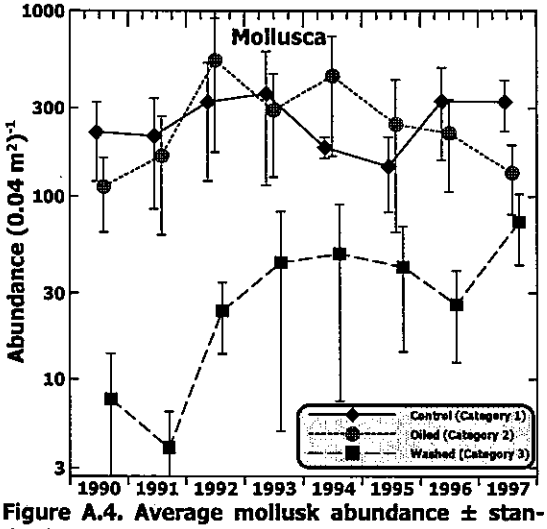

Figure A.4. Average mollusk abundance ± standard error (Figure 12)

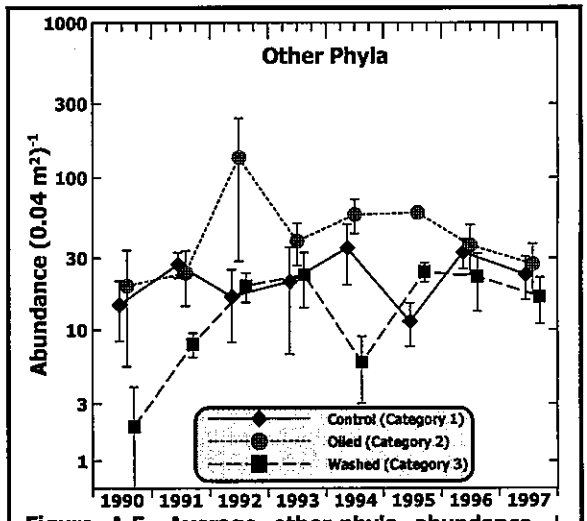

Figure A.5. Average other-phyla abundance ± standard error (Figure 13)

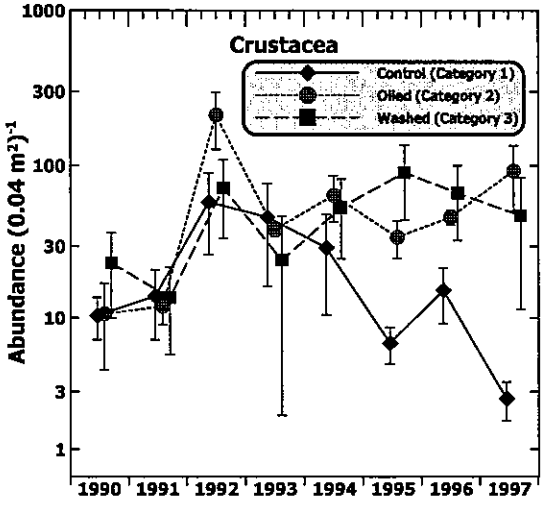

Figure A.6. Average crustacean abundance ± standard error (Figure 14)

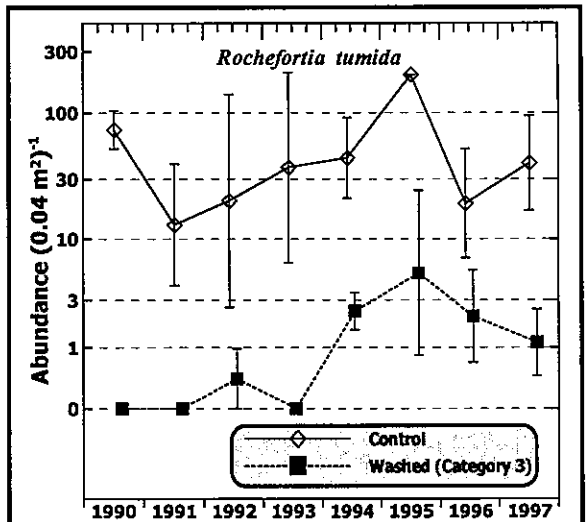

Figure A.7. Average *Rochefortia tumida* abundance ± standard error (Figure 17)

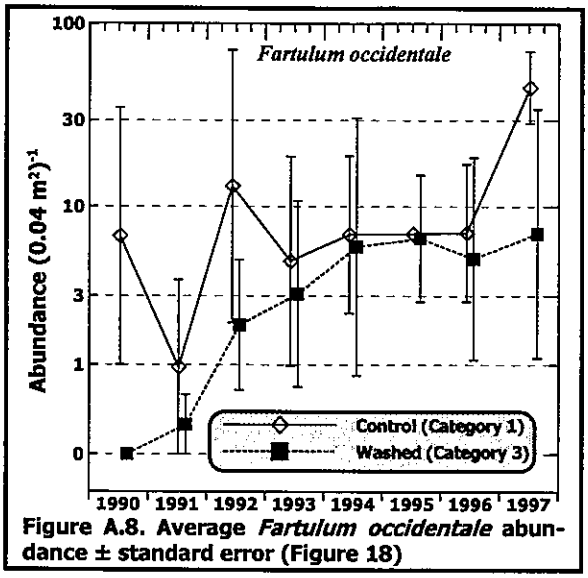

Figure A.8. Average *Fartulum occidentale* abundance ± standard error (Figure 18)

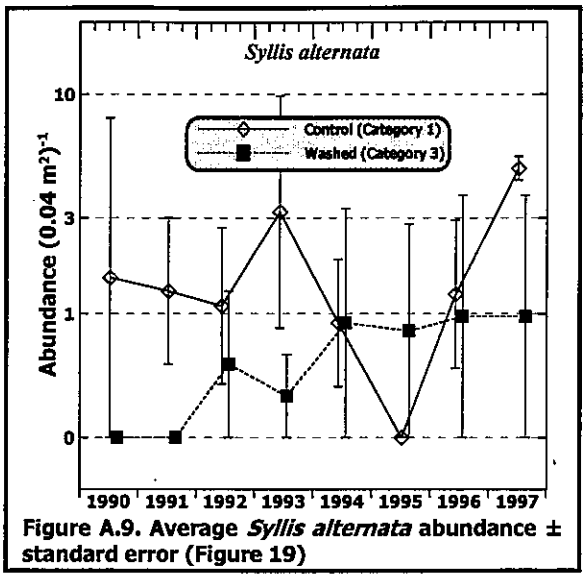

Figure A.9. Average *Syllis alternata* abundance ± standard error (Figure 19)

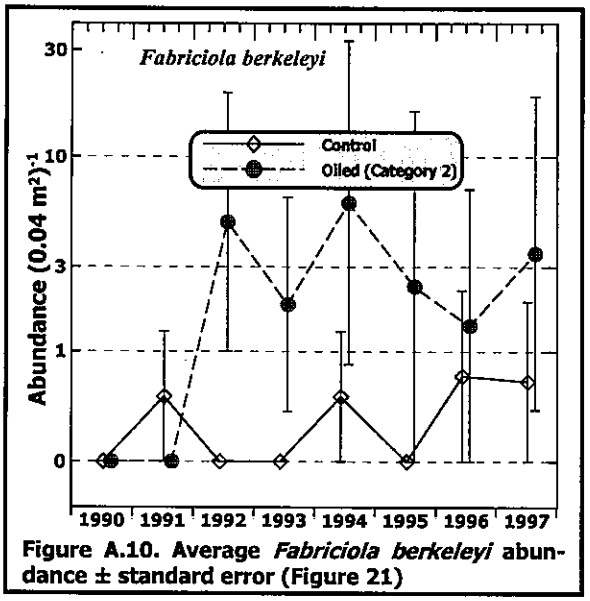

Figure A.10. Average *Fabriciola berkeleyi* abundance ± standard error (Figure 21)

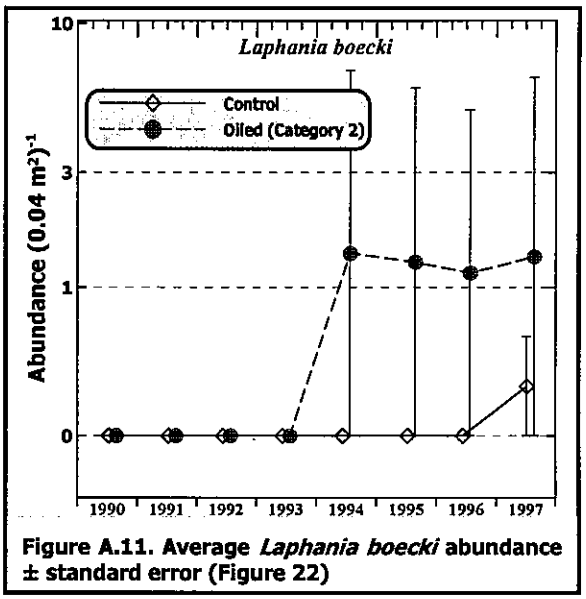

Figure A.11. Average *Laphania boecki* abundance ± standard error (Figure 22)

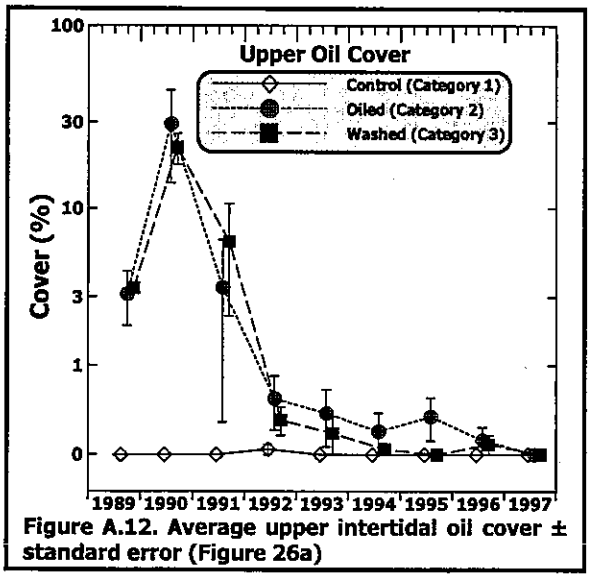

Figure A.12. Average upper intertidal oil cover ± standard error (Figure 26a)

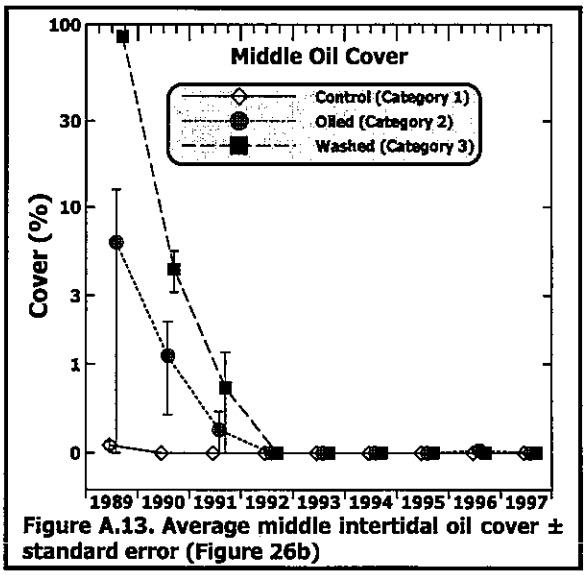

Figure A.13. Average middle intertidal oil cover ± standard error (Figure 26b)

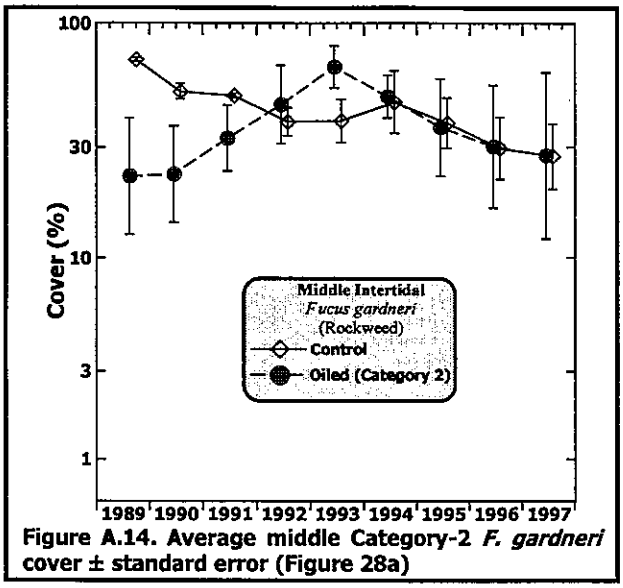

Figure A.14. Average middle Category-2 *F. gardneri* cover ± standard error (Figure 28a)

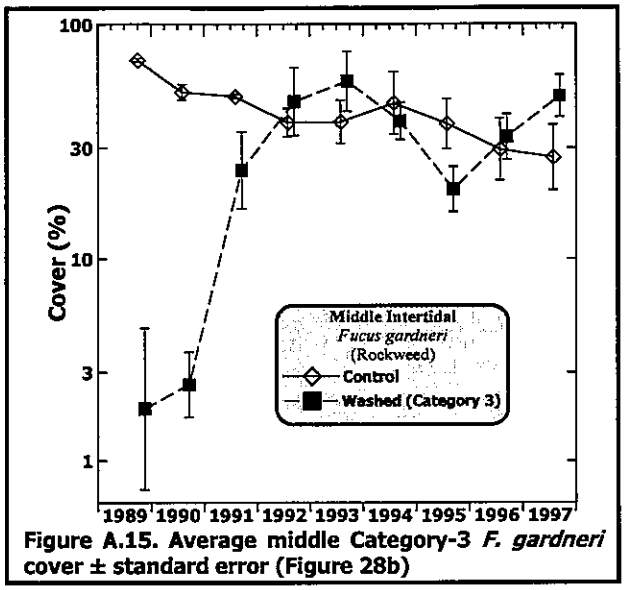

Figure A.15. Average middle Category-3 *F. gardneri* cover ± standard error (Figure 28b)

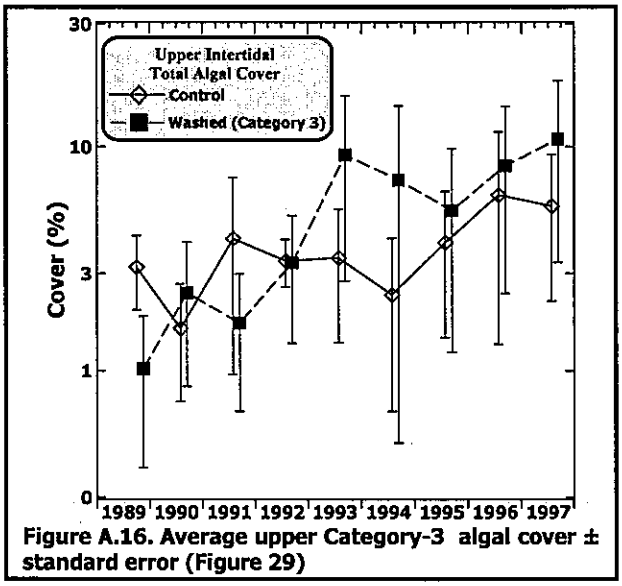

Figure A.16. Average upper Category-3 algal cover ± standard error (Figure 29)

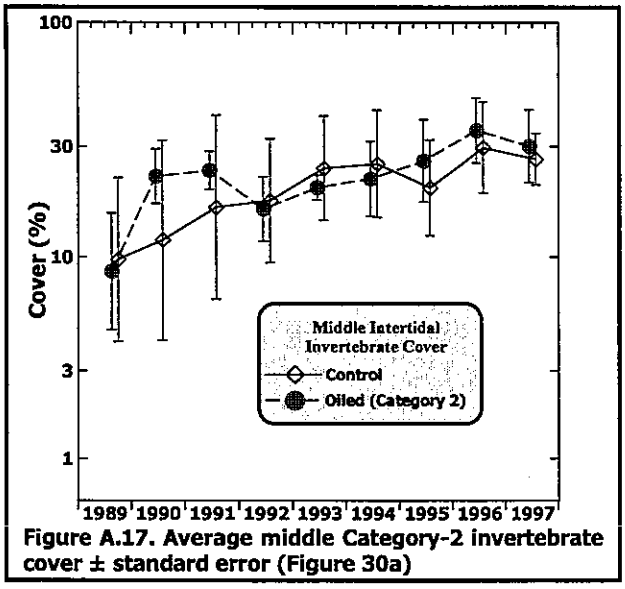

Figure A.17. Average middle Category-2 invertebrate cover ± standard error (Figure 30a)

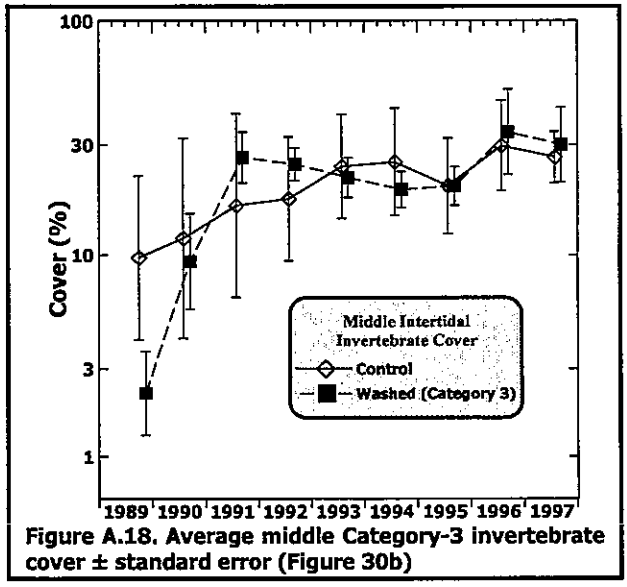

Figure A.18. Average middle Category-3 invertebrate cover ± standard error (Figure 30b)

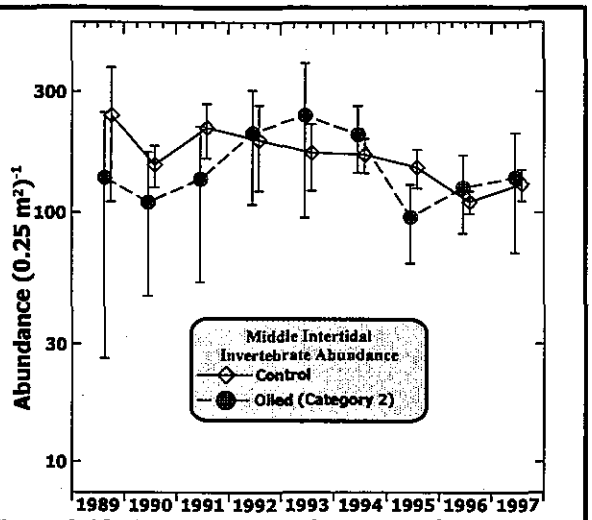

Figure A.19. Average middle Category-2 invertebrate abundance ± standard error (Figure 31a)

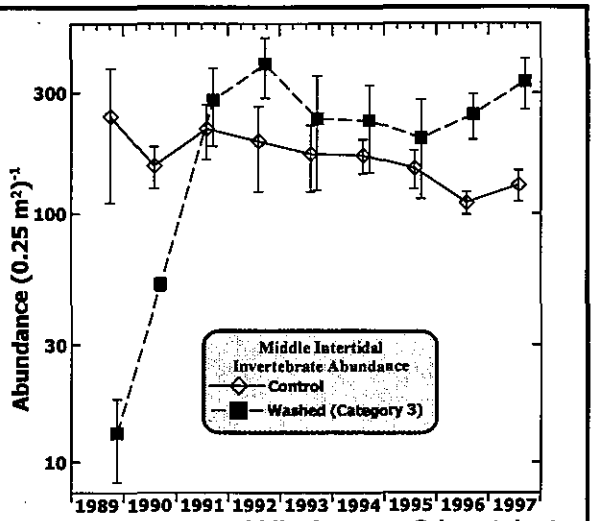

Figure A.20. Average middle Category-3 invertebrate abundance ± standard error (Figure 31b)

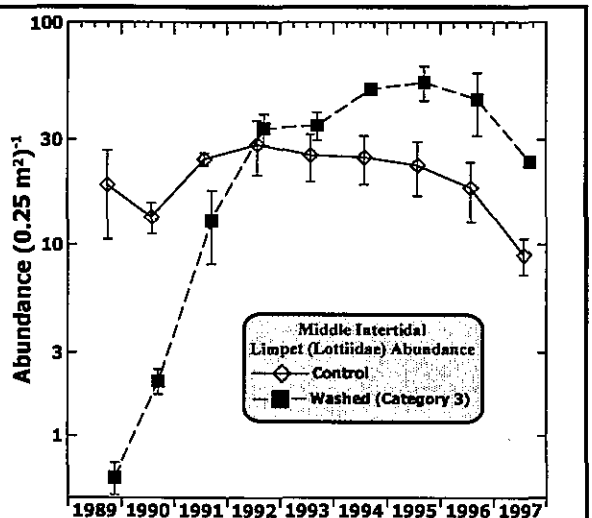

Figure A.21. Average middle Category-3 limpet abundance ± standard error (Figure 32a)

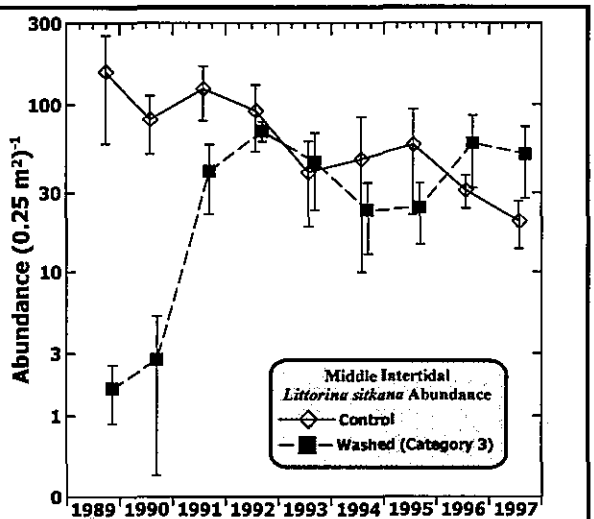

Figure A.22. Average middle Category-3 *L. sitkana* abundance ± standard error (Figure 32b)

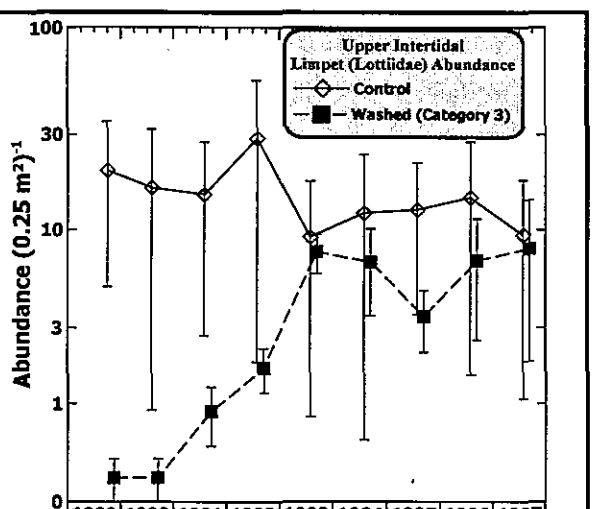

Figure A.23. Average upper Category-3 limpet abundance ± standard error (Figure 34a)

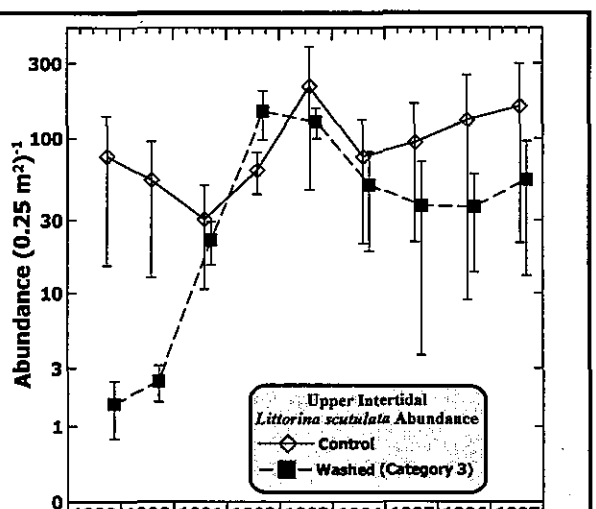

Figure A.24. Average upper Category-3 *L. scutulata* abundance ± standard error (Figure 34b)

A.5 GRAIN SIZE COMPARABILITY

The methodology for determining sediment particle size was changed in 1997. The new method is based on standard methods outlined by Plumb (1981) where dry-sieved size fractions are weighed and the remaining mud fraction is determined by pipette analysis. Grain-size in samples collected prior to 1997 was determined volumetrically after wet sieving as described by McNeil and Ahnell (1964). The methodology was changed to conform to currently accepted procedures for processing sediment samples and to increase the accuracy of the fine-particulate determinations. However, the change in grain-size methodology introduces artificial temporal variability into the database of sediment parameters, which could be incorrectly ascribed to changes in the physical environment within Prince William Sound. Hence, a comparability study was conducted on the sediment samples collected in 1997 to determine the extent of these method-related differences. This sub-appendix describes the results of that comparability study and also described modifications made to the standard method of Plumb (1981) to accommodate marine sediments.

Fine Sediments

In both methodologies, determination of particle-size distribution within a sediment sample is a two-step procedure. Coarser fractions, including sands and gravel, are determined from direct measurement of the displacement volume or the weight of fractions separated after wet or dry sieving. The fine fraction, which includes silt and clay, is determined indirectly. In samples collected prior to 1997, this mud fraction was determined from the volume of settling-tube accumulations. The mud fraction in samples collected in 1997 was determined by the pipette method, which was modified as described below. The pipette method is one of the most commonly used techniques used in the analysis of fine particles. It is based on the differential settling rate of sediments in suspension as described by Stoke's Law. Sediment suspensions are extracted by pipette from a settling column at given depths and times to collect particulates of a specific maximum dimension. Drying and weighing of the extract provides an accurate extension of the grain-size distribution determined by weighing coarser sediments collected after dry sieving.

In contrast to the pipette method, the volumetric determination of settled fine sediments used in samples collected prior to 1997, is not compatible with displacement volumes measured in coarse sediments after wet sieving. Because the volume of settled fine particulates contains interstitial water and displacement volume does not, the fraction of mud is overestimated relative to coarser sediments. This explains the occurrence of an anomalous second mode observed in the grain-size distributions determined prior to 1997. The second fine-fraction mode is evident in the volumetric grain-size distribution shown in Figure 9 in Chapter 2. McCave and Syvitski (1991)

summarize other limitations associated with volumetric determination in settling tubes: "*However, the rather imprecise choice of bulk density values, wall-effect problems in the lower portion of the tube, operator dependence, and low resolution of results have made this class of tubes obsolete, although their cost is very low.*" Because of these limitations and their tendency to overestimate fine sediments relative to coarse fractions, the mud fraction was excluded from multivariate analyses relating environmental factors to infaunal communities.

Coarse Sediments

In contrast to mud, the two methodologies result in comparable sand and gravel distributions, at least within the precision of the volumetric determinations. Theoretically, particulate distributions determined from volume or mass should be equivalent if the measurements are equally precise and mineral densities are not a function of size. The method-comparability study performed on sediments collected in 1997 assessed the degree to which mineral density was consistent across size fractions. Normally, heavy minerals tend to preferentially occur in the finer sieve classes (Matthews, 1991). However, little density-related bias was observed in the comparability study. Also, microscopic examination indicated that the mineralogy of sand particulates was similar to that of gravel and pebbles. Although the color, strike plane, and grain-form (shape) varied widely among sites, the mineralogy collected from a single site was qualitatively consistent across particulate size.

This qualitative consistency in mineralogy was confirmed with quantitative measurements of mineral density Figure A.25. The figure shows that the relationship between volume and mass measurements was generally consistent among the 89 samples tested in the comparability study. The resulting mean density of 2.66 g/mL is typical of rock-forming silicate minerals (Kranck and Milligan, 1991). The largest deviations generally occur at small displacement volume (< 1 mL) which approaches the measurement error. The measurement error for volumetric determinations is more than an order of magnitude larger than for weight measurements.

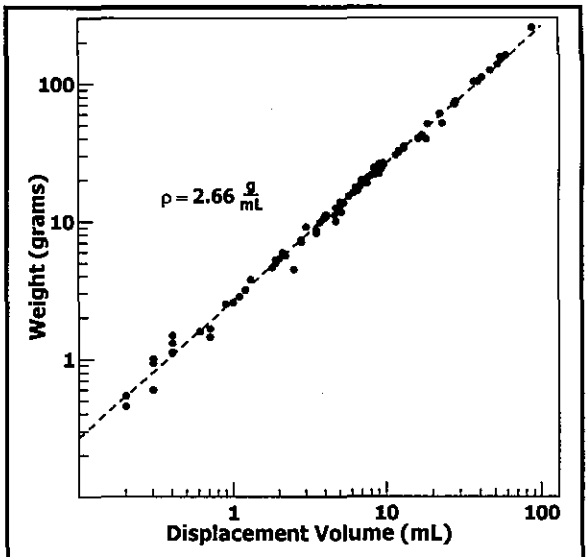

Figure A.25. Mineral density determined from contemporaneous measurements of mass and volume across a wide range of size fractions

The parallel analyses conducted as part of the comparability study reveal the imprecision in volume measurements. First, the sediment samples were processed using the standard dry-sieve technique of Plumb (1981). The weight of each fraction was determined to within ±0.001 g and total sample mass ranged between 200 g and 300 g. Next, displacement volume of each dry-sieved fraction was determined by placing it into a graduated cylinder with a pre-measured volume of water. The increase in water level indicated the displacement volume of the particulates in that fraction as described by McNeil and Ahnell (1964). For the fractions with low sediment volume, a 10-mL graduated cylinder was used to determine volume to within ±0.1 mL. Use of smaller cylinders was impractical. Even with a 10-mL cylinder, great care was required to keep sediment from adhering to the side and to eliminate bubbles that were entrained in submerged sediment. Although total sediment volume approached 100 mL at most sites, the fine sediment volume was less than 4 mL. Consequently, the determination of fine-sand volume was subject to a large measurement error that typically exceeded several percent of the observation. Although increasing the total sample size could improve the relative error of volumetric determinations, practical limitations suggest that it would never achieve the precision (≈±0.1%) obtained by weighing small samples of fine sands.

A slight difference in the cumulative distributions determined by the two measurement techniques was revealed by the comparability study (Figure A.26). Cumulative sediment volume was overestimated by about 1% to 2% when cumulative volume exceeded 90%. Correspondingly, the volume of the fine and medium sands that constitute less than 10% of the sample was underestimated compared to fractions determined by dry weight. This bias could be a consequence of small size-related differences in mineral density unresolved in Figure A.25 or could be introduced by difficulties in measuring small displacement volumes, or both.

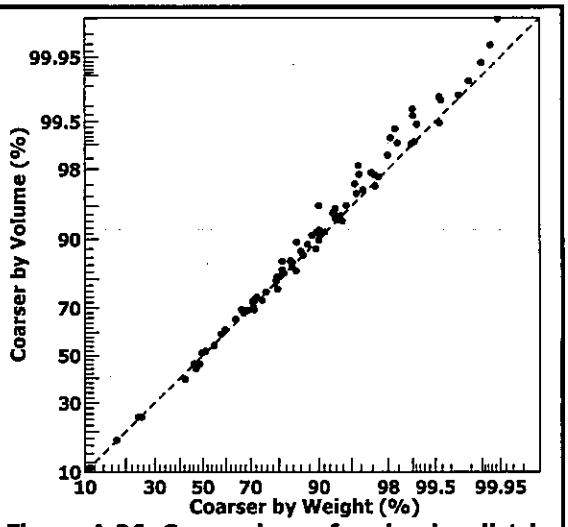

Figure A.26. Comparison of grain-size distribution determined volumetrically and by weight for all 1997 samples

In any regard, the effect of the bias is small compared to differences in the distributions among sites. The representative distributions in Figure A.27 all show the methodological deviations indicated in Figure A.26. However, the 1% to 2% overestimate in cumulative coarse fractions is

small compared to the differences in the shape and level of the distributions between sites. These spatial differences in environmental factors are the focus of the canonical correspondence analysis described in Chapter 2. For this purpose, the two methods for determining coarse sediment fractions can be considered comparable and historical volumetric measurements were combined with the more accurate determinations made in 1997. In contrast, fundamental methodological differences in the determination of the mud fraction preclude conjoining those observations in the database.

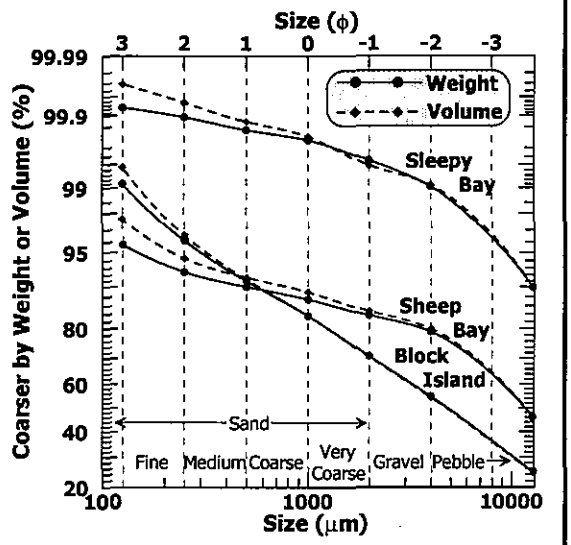

Figure A.27. Cumulative probability distribution of grain-size determined volumetrically and by weight in representative samples collected in 1997

Modification of the Pipette Method

Despite limitations in historical estimates, accurate determination of fine fractions will be important in continued intertidal monitoring of Prince William Sound. Because of their size, infauna are more sensitive to the finer sediment fractions. In addition, the entire particle-size distribution is used to infer the influence of depositional processes (Kranck and Milligan, 1991). These aspects are important for assessing whether there is ongoing infaunal recovery at washed (Category-3) sites that is linked to redeposition of fine sediments as postulated in Chapter 2. To obtain accurate estimates of the mud fraction in marine sediments, it is necessary to modify the standard pipette method outlined by Plumb (1981). Specifically, the weight of the solid content left after evaporation of the pipette extraction requires correction for dissolved salts. The refined procedure, which is described below, was applied to the 1997 intertidal samples and produced a significantly more accurate estimate of fine grain-size fraction.

During wet-sieving of sediment samples, dissolved salt within interstitial water washes into the graduated cylinder used in the pipette analysis. The presence of these accumulated dissolved solids will increase the apparent weight of the fine fraction unless they are accounted for analytically. The corrected weight of the fine fraction (W_f) can be determined from salinity and moisture-content measurements and is given by:

$$W_f = W_p - \frac{M \cdot S \cdot W_w}{1000}$$

where:

W_p = dry weight (mg) of the fine fraction after accounting for peptizing agent and pipette fraction,

M = moisture content (%) of the original sediment sample as collected in the field,

S = salinity (‰) within the interstitial water from the original field sample, and

W_w = wet weight (mg) of the sample used in the grain-size analysis.

The moisture content (M) was determined by drying a subsample from the original field sample. Salinity measured in nearby surface waters acted as a proxy for salt concentrations within interstitial waters. Because there are numerous freshwater sources along the shoreline of Prince William Sound, salinity was found to vary substantially among the sites. It is likely that salinity also varies over smaller spatial scales, namely within sites, and it is recommended that future estimates of interstitial salinity be determined from a conductivity probe inserted directly in the sample. This would not only improve the dissolved-solid correction term for grain-size determination, but would provide another important environmental parameter (*viz.*, salinity) that probably has a significant influence on site-specific infaunal communities.

Appendix B.1 Mean density (m^{-2}) of enumerated organisms collected at infaunal monitoring sites during 1997

Taxon	Bainbridge Bight	Block Island	Crab Bay	Herring Bay	Mussel Beach	NW Bay	Outside Bay	Sheep Bay	Shelter Bay	Sleepy Bay	Snug Harbor
Abarenicola	92.4	—	—	—	—	—	—	—	—	—	—
Acarina	23.1	184.8	—	438.8	—	—	—	—	—	161.7	—
Acteocina	—	—	—	—	—	—	—	23.1	—	69.3	—
Aeolidiacea	—	—	—	—	23.1	—	—	—	—	—	—
Allorchestes angusta	—	—	—	—	—	—	—	—	—	—	23.1
Alvania compacta	—	415.8	—	184.8	877.7	923.9	1201.1	785.3	277.2	23.1	207.9
Ameira longipes	—	—	—	—	—	—	—	—	23.1	—	—
Amphiascopsis cinctus	46.2	—	—	—	23.1	23.1	115.5	—	—	23.1	46.2
Ampithoe	—	—	—	92.4	—	—	—	—	92.4	—	—
Anomura	—	—	—	23.1	—	—	—	23.1	—	—	—
Anthopleura artemisia	—	—	23.1	—	—	—	—	—	—	—	—
Aphelochaeta	—	—	—	—	—	—	1385.8	—	—	—	—
Apodida	—	—	—	23.1	—	—	—	—	—	23.1	—
Araneae	—	—	—	—	—	—	23.1	—	—	—	—
Aricidea	—	—	—	—	—	—	23.1	—	—	—	—
Armandia brevis	—	231	—	23.1	23.1	—	508.1	231	—	23.1	—
Asabellides sibirica	—	46.2	—	—	—	—	—	—	—	—	—
Ascidiacea	—	—	—	—	—	—	—	—	—	—	23.1
Asteroidea	—	—	—	—	—	—	23.1	—	—	—	69.3
Balanus	—	—	—	—	—	—	—	—	46.2	—	—
Balanus crenatus	—	—	—	—	—	—	—	—	69.3	—	—
Barantolla americana	23.1	115.5	92.4	184.8	23.1	—	23.1	23.1	23.1	—	23.1
Bittium	—	—	—	—	23.1	—	—	—	—	—	115.5
Bivalvia	—	92.4	—	23.1	—	23.1	—	23.1	231	46.2	—
Brada sachalina	23.1	—	—	—	—	—	—	—	—	—	—
Cancer oregonensis	—	—	—	—	23.1	—	—	—	—	—	—
Capitella capitata complex	115.5	—	—	—	—	—	69.3	—	—	—	—
Chiridota	—	—	—	—	—	—	23.1	—	—	—	23.1
Chironomidae	—	—	—	323.4	69.3	—	69.3	—	23.1	—	231
Cingula	7044.7	—	23.1	785.3	—	369.6	—	—	—	—	—
Cirratulus spectabilis	—	—	—	—	184.8	—	92.4	—	—	69.3	115.5
Clinocardium	—	—	—	23.1	—	—	—	—	—	—	—
Coleoptera	—	—	23.1	—	—	—	—	—	—	—	—

Appendix B.1 Mean density (m^{-2}) of enumerated organisms collected at infaunal monitoring sites during 1997
(Continued)

Taxon	Bainbridge Bight	Block Island	Crab Bay	Herring Bay	Mussel Beach	NW Bay	Outside Bay	Sheep Bay	Shelter Bay	Sleepy Bay	Snug Harbor
Collembola	—	—	—	—	46.2	92.4	—	—	46.2	692.9	46.2
Corophium	—	—	—	461.9	—	—	23.1	23.1	—	—	115.5
Corophium brevis	—	—	—	392.7	—	—	—	—	—	—	—
Cranopsis cucullata	—	—	—	—	23.1	—	—	—	—	—	—
Cryptomya californica	23.1	—	—	—	—	—	23.1	—	—	—	46.2
Cumella vulgaris	23.1	184.8	1085.6	1293.4	—	23.1	—	—	369.6	—	323.4
Cyclopoida	—	—	—	—	—	—	—	—	—	23.1	—
Cyclopoida poecilostomatoida	—	—	—	—	—	—	—	—	—	23.1	—
Dactylopusia copepodid	23.1	23.1	—	—	92.4	—	—	—	23.1	23.1	46.2
Dendrochirotida	—	—	—	—	—	—	—	—	—	—	23.1
Desdemelita californica	—	—	—	—	115.5	—	23.1	—	—	115.5	—
Diplodonta aleutica	—	—	—	—	92.4	—	—	—	—	—	—
Echinolaophonte	—	46.2	—	—	—	—	646.7	—	—	—	—
Echiurus echiurus alaskanus	69.3	—	—	23.1	—	—	—	—	—	—	23.1
Epicaridea	—	—	—	—	—	—	—	—	23.1	—	—
Eteone	69.3	46.2	—	161.7	970.1	92.4	69.3	46.2	46.2	115.5	415.8
Eusyllis	277.2	—	—	—	—	—	—	—	—	—	—
Exogone dwisula	—	115.5	—	—	—	—	—	—	—	—	—
Fabriciola berkeleyi	115.5	—	—	23.1	—	23.1	—	—	—	231	1963.3
Fartulum occidentale	1293.4	23.1	—	46.2	1917.1	184.8	1986.4	438.8	—	2309.7	531.2
Gammaridea	—	—	—	23.1	46.2	—	—	—	—	46.2	—
Gastropoda	—	46.2	—	46.2	23.1	—	—	207.9	23.1	69.3	—
Glycera nana	—	46.2	—	—	23.1	—	46.2	23.1	—	92.4	—
Glycinde polygnatha	—	23.1	—	—	—	—	—	231	—	—	—
Gnorimosphaeroma oregonensis	—	—	69.3	—	—	—	—	—	369.6	392.7	—
Halacaridae	231	46.2	—	—	23.1	—	46.2	23.1	—	—	531.2
Halectinosoma	—	—	—	—	—	—	23.1	—	—	—	—
Harmothoe imbricata	115.5	—	—	69.3	—	—	69.3	254.1	—	—	—
Harpacticoida	277.2	2078.8	—	3764.9	—	—	300.3	46.2	69.3	923.9	69.3
Harpacticus uniremis	1039.4	—	23.1	—	69.3	—	762.2	—	—	23.1	—
Hemigrapsus nudus	—	—	—	—	—	—	—	323.4	—	—	—
Hemigrapsus oregonensis	—	—	—	—	—	—	—	92.4	—	—	—
Hesionidae	—	—	—	—	—	—	23.1	—	—	—	—
Heterolaophonte discophora	—	—	—	—	—	115.5	346.5	—	46.2	—	438.8
Hiatella arctica	92.4	369.6	—	485	46.2	300.3	46.2	138.6	900.8	3903.4	—
Ianiropsis	—	—	—	—	—	—	—	—	415.8	—	—

Appendix B.1 Mean density (m^{-2}) of enumerated organisms collected at infaunal monitoring sites during 1997
(Continued)

Taxon	Bainbridge Bight	Block Island	Crab Bay	Herring Bay	Mussel Beach	NW Bay	Outside Bay	Sheep Bay	Shelter Bay	Sleepy Bay	Snug Harbor
Ianiropsis kincaidi	—	23.1	—	—	—	—	—	—	161.7	—	323.4
Idotea	—	—	—	—	—	—	—	23.1	—	—	—
Insecta	46.2	—	23.1	—	—	—	—	—	—	—	—
Lacuna	—	—	—	—	23.1	—	23.1	46.2	600.5	531.2	—
Laophonte cornuta	—	—	—	—	—	23.1	138.6	—	—	—	92.4
Laphania boecki	—	—	—	—	600.5	—	—	23.1	—	—	—
Leptochelia savignyi	—	—	—	69.3	69.3	—	23.1	—	—	—	2032.6
Leptosynapta	—	46.2	—	—	23.1	—	115.5	69.3	—	—	—
Macoma	23.1	323.4	—	184.8	—	—	115.5	—	—	—	—
Macoma balthica	—	—	92.4	69.3	—	—	—	—	—	—	—
Macoma inquinata	115.5	277.2	—	—	46.2	—	161.7	92.4	—	—	—
Macoma nasuta	23.1	—	—	23.1	—	—	69.3	—	—	—	—
Margarites marginatus	—	—	—	—	—	—	—	—	—	69.3	69.3
Margarites pupillus	—	—	—	—	—	—	23.1	—	—	23.1	—
Mediomastus californiensis	—	—	—	—	23.1	—	—	—	—	—	—
Megamphopus	—	—	—	—	—	—	—	—	—	46.2	—
Modiolus	—	23.1	—	23.1	46.2	—	115.5	—	—	69.3	—
Mya arenaria	—	23.1	—	—	—	—	—	—	—	—	—
Naineris quadricuspida	1940.2	—	—	—	115.5	—	—	—	—	—	—
Natica clausa	46.2	—	—	—	—	—	—	—	—	—	—
Nematoda	1131.8	2933.4	692.9	6582.7	2332.8	2009.5	4734.9	2194.2	3395.3	3025.7	993.2
Nemertea	300.3	438.8	369.6	461.9	877.7	623.6	161.7	785.3	161.7	369.6	254.1
Nereis	—	—	—	254.1	—	—	—	—	—	—	—
Nereis vexillosa	46.2	—	—	—	—	—	—	—	115.5	—	—
Nerilla digitata	—	—	23.1	—	—	—	—	—	—	—	—
Odostomia	23.1	115.5	—	—	46.2	—	184.8	392.7	23.1	92.4	—
Oligochaeta	2563.8	415.8	2910.3	7067.8	2009.5	1755.4	254.1	323.4	4042	161.7	254.1
Ophelia limacina	—	—	—	—	92.4	—	92.4	23.1	—	115.5	—
Orbiniella nuda	1178	—	—	—	—	—	—	—	—	—	—
Orobitella	—	—	—	—	46.2	—	—	—	—	—	—
Ostracoda	—	—	23.1	323.4	—	—	—	—	—	69.3	—
Owenia fusiformis	—	46.2	—	23.1	—	—	23.1	577.4	—	138.6	—
Paguridae	—	—	—	—	—	—	—	46.2	69.3	—	—
Pagurus	—	—	—	—	—	—	—	—	69.3	—	—
Paramoera	23.1	—	—	923.9	—	—	—	—	1686.1	115.5	138.6

Appendix B.1 Mean density (m^{-2}) of enumerated organisms collected at infaunal monitoring sites during 1997
(Continued)

Taxon	Bainbridge Bight	Block Island	Crab Bay	Herring Bay	Mussel Beach	NW Bay	Outside Bay	Sheep Bay	Shelter Bay	Sleepy Bay	Snug Harbor
Pectinaria granulata	23.1	692.9	—	46.2	—	—	46.2	207.9	—	—	—
Pentamera	—	—	—	—	—	—	—	23.1	—	—	—
Phascolosoma agassizi	23.1	23.1	—	46.2	161.7	—	23.1	—	—	—	—
Pholoe minuta	692.9	323.4	—	115.5	831.5	23.1	277.2	184.8	23.1	207.9	392.7
Pisces	23.1	—	—	—	46.2	—	—	—	—	46.2	—
Platynereis bicanaliculata	—	—	—	—	—	—	—	—	—	46.2	—
Pleustidae	46.2	—	—	—	—	—	—	—	—	115.5	—
Podocopa	23.1	—	23.1	—	—	46.2	23.1	—	138.6	46.2	69.3
Polycirrus	—	—	—	—	—	—	—	—	—	—	23.1
Polydora cardalia	—	—	—	—	—	—	—	23.1	—	—	—
Polydora caulleryi	—	23.1	—	—	—	—	46.2	46.2	—	23.1	—
Polydora limicola	23.1	—	—	—	—	—	—	—	—	—	—
Polydora quadrilobata	23.1	46.2	—	—	—	—	184.8	—	—	—	—
Polygordius	—	—	—	—	—	—	—	—	—	23.1	—
Pontogeneia	—	—	—	—	23.1	—	—	—	—	—	—
Pontogeneia ivanovi	—	—	—	—	—	—	—	—	—	115.5	—
Prionospio	—	—	—	—	23.1	—	—	—	—	23.1	—
Prionospio cirrifera	—	—	—	—	—	—	531.2	—	—	—	—
Prionospio jubata	—	—	—	—	—	—	—	254.1	—	—	—
Protodorvillea gracilis	—	—	—	—	—	—	—	138.6	—	—	—
Protothaca staminea	23.1	554.3	92.4	46.2	508.1	138.6	415.8	600.5	23.1	—	46.2
Pseudonychocamptus	—	—	—	—	—	—	23.1	—	23.1	—	23.1
Pygospio elegans	—	—	—	—	—	—	—	92.4	—	23.1	138.6
Rissoidae	1478.2	—	—	1547.5	—	184.8	—	—	69.3	161.7	—
Rochfortia tumida	277.2	623.6	—	92.4	1547.5	46.2	4850.4	577.4	—	69.3	23.1
Saccocirrus eroticus	—	—	—	—	—	92.4	—	—	—	1062.5	—
Saxidomus giganteus	—	138.6	—	—	184.8	23.1	92.4	23.1	—	—	—
Scolelepis squamata	—	—	—	—	—	—	—	—	—	23.1	—
Scoloplos	—	—	—	—	—	—	277.2	—	—	—	—
Sphaerosyllis	—	—	—	—	—	—	46.2	—	—	—	—
Sphaerosyllis californiensis	—	—	—	—	993.2	—	—	—	—	23.1	—
Spinulogammarus subcarinatus	—	—	—	—	—	—	—	—	—	23.1	—
Spio	—	23.1	—	—	—	—	23.1	—	—	69.3	—
Spio filicornis	46.2	—	—	—	—	—	—	—	—	—	—
Staphylinidae	—	—	23.1	—	—	—	—	—	—	—	—
Stichaeidae	—	—	—	—	23.1	—	—	—	—	—	—

Appendix B.1 Mean density (m^{-2}) of enumerated organisms collected at infaunal monitoring sites during 1997
(Continued)

Taxon	Bainbridge Bight	Block Island	Crab Bay	Herring Bay	Mussel Beach	NW Bay	Outside Bay	Sheep Bay	Shelter Bay	Sleepy Bay	Snug Harbor
Strongylocentrotus droebachiensis	23.1	46.2	—	—	—	—	23.1	—	—	—	—
Syllides reishi	69.3	—	—	—	—	—	—	—	—	—	—
Syllis alternata	138.6	877.7	—	—	531.2	—	115.5	92.4	—	277.2	23.1
Telmessus cheiragonus	—	—	—	23.1	—	—	—	—	—	—	—
Turbellaria	23.1	—	—	—	—	—	—	—	—	—	—
Turtonia minuta	—	—	—	—	—	—	69.3	—	—	—	—
Typhlamphiascus	—	—	—	—	—	—	69.3	—	—	—	—
Volutopsiu	—	46.2	—	—	—	—	—	—	—	—	—

Appendix B.2 Physicochemical properties of composite sediment samples collected at infaunal monitoring sites during 1997

Location	Fraction (%) by Weight within Specified Particle-Diameter Range										TKN (mg-N/kg)	TOC (%)
	> 12.7 mm	12.7 to 4.0 mm	4.0 to 2.0 mm	2.0 to 1.0 mm	1.0 to 0.5 mm	500 to 250 µm	250 to 125 µm	125 to 62.5 µm	62.5 to 2 µm	<2 µm		
Bainbridge Bight	10.90	69.47	13.36	3.56	0.83	0.47	0.57	0.50	0.33	0.00	163	0.34
Block Island	25.13	29.80	16.05	12.65	7.47	5.08	2.94	0.54	0.32	0.03	65	0.25
Crab Bay	51.12	12.89	8.22	8.56	7.70	7.24	1.54	0.41	1.43	0.89	248	0.66
Herring Bay	42.21	29.07	8.10	5.32	4.70	4.46	3.32	1.83	0.99	0.00	138	0.50
Mussel Beach South	66.41	14.41	3.99	5.09	4.50	3.30	1.20	0.47	0.51	0.13	446	0.57
NW Bay West Arm	46.94	21.82	10.21	7.48	6.78	5.71	0.61	0.19	0.25	0.00	198	0.44
Outside Bay	17.38	40.65	12.68	12.47	6.26	3.01	2.90	2.85	1.67	0.13	394	0.47
Sheep Bay	46.13	32.92	4.67	3.81	2.50	2.51	3.27	2.23	1.93	0.04	351	0.51
Shelter Bay	48.67	18.68	8.31	5.71	4.66	11.52	2.19	0.25	0.01	0.00	298	0.24
Sleepy Bay	89.91	9.13	0.52	0.21	0.07	0.06	0.03	0.02	0.05	0.00	216	0.40
Snug Harbor	74.23	16.07	2.55	1.5	1.19	1.35	1.22	0.75	0.95	0.19	82	0.31
Zaikof	24.13	25.42	10.06	11.44	8.28	13.48	5.52	0.79	0.77	0.10	538	0.44

Appendix C.1 Mean epibiotic population and substrate cover along upper intertidal transects during 1997

Taxon	NW Bay Islet	Snug Harbor	Hogg Bay	Mussel Beach North	Eshamy Bay	Herring Bay	Bass Harbor	Outside Bay	Block Island	Mussel Beach South	Elrington W Islet E	Elrington W Islet N	Elrington W Islet W
Algal Cover (%)													
Fucus gardneri	-	1.8	3.8	27.2	-	0.6	3.7	0.9	20.3	5.6	0.7	17.1	10.7
Encrusting brown algae	39.0	-	-	-	-	-	-	-	0.7	-	-	33.0	-
Hildenbrandia rubra	0.1	1.0	0.1	1.9	0.1	0.4	0.3	-	0.3	3.3	0.2	17.2	20.6
Gloiopeltis furcata	-	0.2	-	1.8	-	-	6.7	4.2	0.2	-	-	0.2	-
encrusting red algae	-	4.0	-	8.1	0.1	-	-	-	-	-	-	-	-
Fucus gardneri (germlings)	0.3	0.7	0.6	1.4	0.3	0.5	-	0.4	1.3	0.9	0.1	0.2	0.3
endozoic green algae	-	0.8	0.1	2.6	0.1	-	2.0	0.4	0.1	0.4	-	0.1	0.2
blue-green algae, crust	-	0.6	-	1.3	-	-	-	0.2	-	-	0.4	2.3	1.1
Neorhodomela oregona	-	-	-	1.8	-	-	-	-	3.0	-	-	-	-
Ralfsia spp.	-	-	-	2.1	-	-	-	0.4	2.0	-	-	-	0.2
Chaetomorpha spp.	-	-	-	0.8	-	-	-	0.3	-	-	-	-	-
Melanosiphon intestinalis	-	-	-	0.6	-	-	-	-	-	-	-	0.1	-
Cladophora sericea	-	-	-	0.6	-	-	-	-	-	-	-	-	-
coralline crustose algae	-	-	-	0.5	-	-	-	-	-	-	-	-	-
Endocladia muricata	-	-	-	-	-	-	-	0.4	-	-	-	-	-
blue-green algae, spheroids	-	-	-	-	-	-	-	-	-	-	-	0.2	-
Enteromorpha intestinalis	-	-	-	-	-	-	-	-	-	-	-	0.1	-
Pilayella littoralis	-	-	-	-	-	-	-	-	-	-	-	0.1	-
Verrucaria spp.	1.7	43.0	95.2	11.2	24.0	0.8	-	-	19.6	1.7	0.1	18.8	9.3
Invertebrate Cover (%)													
Semibalanus balanoides	0.2	2.1	0.1	1.9	0.2	0.4	42.0	-	4.7	0.1	15.7	0.1	1.0
Chthamalus dalli	0.2	0.2	0.1	7.3	0.3	0.3	2.6	2.6	-	0.1	1.6	2.0	0.1
Balanus/Semibalanus spp. (set)	0.1	0.2	-	-	0.1	-	16.0	-	-	-	0.4	0.3	0.2
Balanus glandula	0.5	0.6	0.3	0.7	0.2	0.1	-	0.9	-	-	0.8	6.0	0.6
Mytilus cf. *trossulus*	0.7	1.1	-	3.0	0.1	0.2	1.5	0.1	1.1	0.3	0.5	0.2	0.1
Chthamalus dalli (set)	-	-	-	1.5	-	-	2.4	0.2	-	-	0.4	-	-
Semibalanus balanoides (set)	-	-	-	1.1	0.2	-	-	-	2.4	-	-	0.3	0.2
Balanus/Semibalanus spp.	-	-	-	-	-	-	-	-	-	1.6	-	-	-
Mytilus cf. *trossulus* (spat)	-	-	-	-	0.1	-	-	-	-	-	-	0.1	0.2
Semibalanus cariosus	-	-	-	0.1	-	-	0.1	0.1	-	-	-	-	-

Appendix C.1 Mean epibiotic population and substrate cover along upper intertidal transects during 1997
(Continued)

Taxon	NW Bay Islet	Snug Harbor	Hogg Bay	Mussel Beach North	Eshamy Bay	Herring Bay	Bass Harbor	Outside Bay	Block Island	Mussel Beach South	Elrington W Islet E	Elrington W Islet N	Elrington W Islet W
Substrate Cover (%)													
rock	99.0	-	100.0	99.5	98.0	84.0	100.0	100.0	100.0	40.0	80.0	100.0	100.0
bare bedrock	65.8	-	3.4	69.2	73.6	83.0	32.6	90.4	64.8	35.0	53.0	21.0	66.0
boulder/cobble	1.0	100.0	-	0.5	2.0	16.0	-	-	-	54.0	20.0	-	-
bare boulder/cobble	1.0	46.0	-	0.5	2.0	16.0	-	-	-	54.0	6.0	-	-
gravel/sand	-	-	-	-	-	-	-	-	-	6.0	-	-	-
bare gravel/sand	-	-	-	-	-	-	-	-	-	6.0	-	-	-
Invertebrate Abundance $(0.25\ m^2)^{-1}$													
Littorina scutulata	136.8	7.8	21.2	20.8	21.4	31.6	437.0	156.8	21.8	4.8	7.6	5.6	
Littorina sitkana	8.0	10.2	70.4	140.1	16.8	10.0	5.8	52.4	51.2	63.4	0.4	3.8	2.8
Mytilus spp. (# live)	11.0	14.6	-	36.5	0.8	1.4	27.4	-	5.4	0.6	8.2	0.2	-
Tectura persona	1.0	8.2	-	6.6	2.2	6.4	5.8	0.2	0.4	20.2	6.2	-	-
Lottia strigatella	-	-	-	0.4	-	-	19.8	3.4	-	-	0.6	4.6	6.8
Mytilus spp. (# live, juv.)	-	-	-	-	1.2	-	-	-	-	-	-	4.6	0.4
Lottiidae (juv.)	0.8	-	-	1.5	-	-	-	-	1.0	-	0.6	1.4	2.6
Lottia pelta	-	-	-	0.7	-	0.2	0.4	-	0.2	-	-	0.4	-
Tectura scutum	-	-	-	-	-	0.4	-	-	-	-	1.2	0.8	1.0
Nucella lamellosa	-	-	-	-	-	-	1.2	0.6	-	-	-	-	-
Siphonaria thersites	-	-	-	-	-	-	0.2	-	-	-	-	1.0	-
Pagurus hirsutiusculus	-	-	-	0.6	-	-	-	-	0.4	-	-	-	-
Ligia spp.	-	0.4	-	0.1	-	-	-	-	-	-	-	-	-
Lottiidae	-	-	-	-	-	-	-	-	0.4	-	-	-	-
Littorina sitkana (juv.)	-	-	-	-	-	-	-	-	-	-	-	-	-
Lottia digitalis	-	-	-	-	-	-	-	-	-	-	-	-	0.4
Modiolus spp.	-	-	-	-	-	-	-	0.1	-	-	-	-	0.2
Nucella lima	-	-	-	0.1	-	-	-	-	-	-	-	-	-
Littorina scutulata (juv.)	-	-	-	-	-	-	-	-	-	-	-	-	-

Appendix C.2 Mean epibiotic population and substrate cover along middle intertidal transects during 1997

Taxon	NW Bay Islet	Snug Harbor	Hogg Bay	Mussel Beach North	Eshamy Bay	Crab Bay	Herring Bay	Outside Bay	Block Island	NW Bay W Arm (Site 52)	NW Bay W Arm (Site 53)	Zaikof Bay
Algal Cover (%)												
Fucus gardneri	45.9	3.3	48.8	20.0	17.2	18.1	64.5	56.5	70.4	35.2	62.0	71.4
Neorhodomela oregona	0.9	0.3	1.5	7.8	3.9	1.9	1.0	0.5	5.1	4.5	2.4	5.8
Hildenbrandia rubra	-	1.4	12.8	0.7	0.9	-	1.4	-	1.3	6.1	3.7	0.2
Fucus gardneri (germlings)	1.7	1.9	3.1	0.6	2.5	1.8	3.5	-	2.2	0.6	0.6	2.3
Cladophora sericea	0.2	0.6	2.9	2.1	1.9	0.8	1.5	6.9	1.6	-	-	2.1
Gloiopeltis furcata	0.6	1.3	2.3	2.6	2.2	2.6	1.7	2.7	0.7	0.6	3.2	-
encrusting brown algae	10.8	-	-	2.8	-	-	-	-	2.0	1.4	-	-
endozoic green algae	1.9	1.2	2.7	<.1	1.5	1.8	1.8	-	2.0	-	0.5	2.2
Chaetomorpha spp.	-	-	-	-	-	-	-	-	-	6.2	7.0	-
blue-green algae, spheroids	0.5	0.7	1.7	<.1	2.0	-	2.9	-	1.1	0.5	0.6	1.0
blue-green algae, crust	-	0.3	-	-	<.1	-	0.5	-	-	5.2	3.7	-
Ralfsia spp.	-	-	0.1	3.5	0.5	0.5	-	1.7	0.5	1.4	-	0.6
Pterosiphonia spp.	-	-	1.8	-	-	-	-	3.3	-	-	1.5	-
Pilayella littoralis	-	0.9	0.7	-	2.0	-	1.6	-	-	-	-	-
Halosaccion glandiforme	-	-	2.3	-	-	-	-	1.9	-	-	0.4	-
coralline crustose algae	-	-	-	2.3	-	-	-	0.3	-	0.8	0.5	-
Melanosiphon intestinalis	0.5	0.4	1.1	0.8	0.3	-	-	-	0.3	0.4	0.1	-
Leathesia difformis	0.5	-	-	1.0	-	-	-	-	0.5	0.9	0.7	-
encrusting red algae	-	2.4	-	<.1	-	-	-	-	0.6	-	-	-
Elachista fucicola	-	-	0.6	-	-	-	-	0.4	-	2.1	-	-
Enteromorpha intestinalis	-	1.5	0.6	-	-	0.5	-	-	-	-	-	-
Soranthera ulvoidea	-	-	<.1	0.6	-	<.1	-	0.5	0.2	0.2	-	0.6
Mastocarpus papillatus	-	0.1	0.8	-	-	-	0.5	0.7	-	-	-	-
Endocladia muricata	-	-	1.9	-	-	-	-	-	-	-	-	-
Palmaria callophylloides	-	-	1.9	-	-	-	-	-	-	-	-	-
Cryptosiphonia woodii	-	-	-	-	-	-	-	1.3	-	0.1	0.1	-
Neorhodomela larix	-	-	-	<.1	-	-	-	-	-	-	1.2	0.1
Iridaea heterocarpa	-	-	-	-	-	-	-	1.1	-	-	-	-
Sphacelaria rigidula	-	-	1.0	-	-	-	-	-	-	-	-	-
Dictyosiphon foeniculaceus	-	-	-	0.6	-	-	-	-	-	-	-	-
Polysiphonia/Pterosiphonia spp.	-	-	-	-	-	-	-	-	-	-	-	0.5
Elachista spp.	-	-	-	-	-	-	-	-	-	-	0.5	-
encrusting green algae	-	-	-	0.3	-	-	-	-	0.1	-	-	-

Appendix C.2 Mean epibiotic population and substrate cover along middle intertidal transects during 1997
(Continued)

Taxon	NW Bay Islet	Snug Harbor	Hogg Bay	Mussel Beach North	Eshamy Bay	Crab Bay	Herring Bay	Outside Bay	Block Island	NW Bay W Arm (Site 52)	NW Bay W Arm (Site 53)	Zaikof Bay
Algal Cover (%) - Continued												
Odonthalia spp.	-	-	-	-	-	-	-	0.3	-	-	-	-
Monostroma grevillei	-	-	0.3	-	-	-	-	-	-	-	-	-
Acrosiphonia spp.	-	-	0.3	-	-	-	-	-	-	-	-	-
Polysiphonia spp.	-	-	0.1	0.2	-	-	-	-	-	-	-	-
Cladophora spp.	-	-	-	-	-	-	-	-	-	-	0.2	-
Odonthalia floccosa	-	-	<.1	-	-	-	-	-	-	-	-	-
Verrucaria spp.	0.3	22.0	0.6	0.5	-	17.5	0.5	-	-	0.3	-	-
Invertebrate Cover (%)												
Semibalanus balanoides	3.3	10.9	3.6	5.5	20.3	3.7	54.5	0.1	19.2	30.4	4.4	2.4
Chthamalus dalli	1.9	1.7	4.1	2.5	2.7	1.4	0.6	10.8	1.5	11.2	8.6	2.7
Balanus glandula	4.5	3.4	3.9	3.0	-	2.9	-	1.6	-	7.2	6.7	11.7
Mytilus cf. *trossulus*	4.1	2.3	<.1	4.3	1.2	6.0	2.0	0.6	9.8	-	-	4.8
Semibalanus cariosus	0.5	2.5	14.1	3.1	-	-	-	1.6	-	0.1	3.7	<.1
Balanus/Semibalanus spp. (set)	0.8	3.1	1.7	0.8	3.2	1.2	1.7	0.6	-	2.3	4.3	2.8
Semibalanus balanoides (set)	-	-	-	1.0	8.5	-	-	-	3.5	-	-	-
Mytilus cf. *trossulus* (spat)	-	-	4.0	0.9	0.8	1.0	1.2	-	-	0.7	3.7	-
Chthamalus dalli (set)	-	-	-	-	-	-	-	1.9	0.1	0.2	-	-
encrusting bryozoan	-	-	-	-	-	-	-	1.0	-	-	-	-
Semibalanus cariosus (set)	-	0.5	-	-	-	-	-	-	-	-	-	-
Spirorbidae	-	-	-	-	-	-	-	0.5	-	-	-	-
Substrate Cover (%)												
rock	89.0	-	38.5	100.0	31.0	35.0	89.0	7.5	100.0	99.4	100.0	87.5
bare bedrock	9.4	100.0	61.5	-	69.0	48.5	11.0	92.5	-	0.6	-	12.5
boulder/cobble	47.6	-	25.0	63.5	31.0	18.6	26.8	6.5	51.0	18.0	36.0	60.4
bare boulder/cobble	-	52.3	14.0	-	29.9	33.6	6.7	64.5	-	0.4	-	12.1
gravel/sand	21.1	-	-	-	-	14.5	-	-	-	-	-	-
bare gravel/sand	-	-	-	-	-	12.3	-	-	-	-	-	-

Appendix C.2 Mean epibiotic population and substrate cover along middle intertidal transects during 1997
(Continued)

Taxon	NW Bay Islet	Snug Harbor	Hogg Bay	Mussel Beach North	Eshamy Bay	Crab Bay	Herring Bay	Outside Bay	Block Island	NW Bay W Arm (Site 52)	NW Bay W Arm (Site 53)	Zaikof Bay
						Invertebrate Abundance $(0.25\ m^2)^{-1}$						
Littorina sitkana	44.5	5.9	8.6	6.4	21.2	30.8	59.9	2.7	93.0	14.4	09.4	43.8
Littorina scutulata	105.9	25.7	26.7	12.8	62.4	10.7	47.6	11.2	36.9	-	44.2	55.8
Mytilus spp. (# live, juv.)	-	-	15.1	09.4	21.2	-	12.5	-	5.0	40.4	04.6	-
Littorina scutulata (juv.)	-	-	1.2	-	-	-	-	-	-	82.0	-	-
Mytilus spp. (# live)	57.2	3.3	0.1	13.3	6.6	48.4	34.3	4.8	16.5	-	-	42.5
Lottiidae (juv.)	20.8	2.2	1.8	45.4	3.0	4.0	7.9	10.3	9.3	12.2	7.4	1.0
Siphonaria thersites	-	-	2.4	0.2	-	0.5	-	53.2	0.2	3.0	22.6	-
Lottia strigatella	2.2	3.6	1.3	2.0	2.4	4.7	3.8	-	6.6	7.6	-	2.4
Lottia pelta	3.7	-	1.8	5.6	0.8	1.6	3.2	2.0	4.5	2.6	9.6	0.2
Nucella lamellosa	-	-	3.8	-	-	-	-	1.5	-	4.6	13.6	-
Littorina sitkana (juv.)	-	-	-	-	-	-	-	-	-	14.4	-	-
Tectura scutum	-	2.8	1.4	0.2	1.9	1.3	0.5	2.8	-	-	-	-
Pagurus hirsutiusculus	1.8	0.2	0.5	2.0	0.9	0.5	0.3	0.3	0.5	1.4	1.0	0.3
Tectura persona	-	5.1	-	-	0.3	0.5	0.4	0.1	-	-	-	3.3
Nucella lima	-	-	-	0.2	-	-	0.7	-	1.9	-	-	1.7
Lottiidae	-	-	-	-	-	-	-	1.2	2.2	-	-	-
Pagurus granosimanus	-	-	-	0.3	-	-	-	-	0.8	-	-	-
Onchidella borealis	-	-	-	-	-	-	-	0.7	-	-	-	-
Hemigrapsus spp.	0.4	0.1	-	-	-	-	-	-	-	-	-	-
Leptasterias hexactis	-	-	-	-	-	-	-	-	0.5	-	-	-
Searlesia dira	-	-	-	0.3	-	-	-	-	0.1	-	-	-
Emplectonema gracile	-	-	-	-	0.1	-	0.1	0.1	-	-	-	-
Amphiporus spp.	-	-	-	-	-	-	-	-	0.1	-	0.2	-
Nucella spp. (juv.)	-	0.3	-	-	-	-	-	-	-	-	-	-
Serpula vermicularis	-	-	0.2	<.1	-	-	-	-	-	-	-	-
Anthopleura artemisia	-	-	-	-	-	-	-	0.2	-	-	-	-
Evasterias troschelii	-	-	-	-	-	-	-	-	-	0.2	-	-
Leptasterias spp.	-	-	0.2	-	-	-	-	-	-	-	-	-
Margarites spp.	-	0.2	-	-	-	-	-	-	-	-	-	-
Gnorimosphaeroma oregonensis	-	-	-	-	-	-	0.1	-	-	-	-	-
Littorina spp. (juv.)	-	-	-	-	0.1	-	-	-	-	-	-	-
Nereidae	-	-	-	-	-	-	-	-	0.1	-	-	-
Paranemertes peregrina	-	-	-	-	-	-	-	-	-	-	-	0.1
Pugettia spp.	-	-	-	-	-	-	0.1	-	-	-	-	-
Pycnopodia helianthoides	-	-	-	-	0.1	-	-	-	-	-	-	-
Anthopleura elegantissima	-	-	-	0.1	-	-	-	-	-	-	-	-
Modiolus spp.	-	-	-	-	-	-	-	-	-	-	-	-